SHORT COMMUNICATION

A broad brush-stroke test of an assumption: does increasing *Callitris* cover reduce native species richness (species density)?

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Abstract: Does an increase in the cover/abundance of *Callitris glaucophylla* or *Callitris endlicheri* affect the number of species recorded in plots (species density) or do other factors such as altitude or logging, fire or grazing history have greater explanatory power? This was tested using survey data from 1351 plots from northern New South Wales. Altitude was found to have the greatest explanatory power in predicting the number of species per plot. Increasing cover/abundance of *Callitris glaucophylla* was found to be positively correlated with increasing species density. Fire was found to have a minor negative effect on species density in *Callitris glaucophylla* stands and grazing a small positive correlation in *Callitris endlicheri* stands.

***Cunninghamia* (2011) 12(2): 115–118**

Introduction

Thirteen *Callitris* species (family Cupressaceae) occur in New South Wales. The two most common are *Callitris glaucophylla* Joy Thomps. & L.A.S. Johnson (White Cypress Pine), a tree, 20–30 m tall when mature that occurs across all mainland Australian states, and *Callitris endlicheri* (Parl.) F.M. Bailey (Black Cypress Pine), a shorter tree (approximately 15–20 m at maturity) that occurs along the eastern mainland states. Both species can form seemingly single-age stands, that continue to become denser to a point where other trees are outcompeted, the stand growth is limited and mortality is low (Lacey 1973; FCNSW 1988).

Some authors have suggested that increasing cover of the dominant *Callitris* reduces the diversity and suppresses the growth of understorey species (Walker *et al.* 1972; Boland *et al.* 1984; Clayton-Green & Ashton 1990). Thinning is frequently carried out on private lands and in state forests to release the overstorey and open up the canopy (Noland 1997) and there is some pressure to continue these practices in lands recently incorporated into National Parks, Nature Reserves and State Conservation Areas. However more recent evidence suggests that a reduction in species richness/density with increasing *Callitris* density does not occur. Andrews (2003) showed that grazing had a greater impact on understorey cover than *Callitris glaucophylla* density, and more recently Thompson and Eldridge (2005a) could find no clear relationships between *Callitris glaucophylla* cover, stem density or wood density and the density, cover or diversity of understorey species; differences were attributed to past disturbances such as forestry practices or grazing rather than increased *Callitris glaucophylla* density.

Here I test whether increased cover of *Callitris glaucophylla* or *Callitris endlicheri* has a measureable effect on species richness (species density) or whether other factors have a greater predictive power, using data from surveyed plots in northern New South Wales.

Methods

The study used existing data recorded from full native floristic 0.04 ha survey sites (20 x 20 m plots) in a 360 km square primarily from the western parts of the Northern Tablelands (west of the New England Highway), Nandewar and Brigalow Belt South Bioregions of New South Wales. Within this area altitude ranges from 50 to 1 210 m above sea level. Sites were chosen if they had some component of either *Callitris glaucophylla* or *Callitris endlicheri* (Fig 1), both species being found across all areas of this region. Two datasets were created, one for each *Callitris* species. Only sites surveyed by the author where chosen in order to limit the error introduced by other surveyors. All data were retrieved from the author's own relational database (Paradox

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12 for windows). A total of 809 sites containing *Callitris glaucophylla* and 542 sites containing *Callitris endlicheri* where chosen.

Within each plot all species present were scored on a modified Braun-Blanquet cover abundance scale (1–6), though the non-native species were excluded from this for the current analyses. Projected cover codes were as follows: 1 <5% few individuals; 2 <5% any number of individuals; 3 = 6–25%; 4 = 26–50%; 5 = 51–75%; 6 >75%. Slope and horizontal elevation were measured using a 'SUUNTO Optical Reading Clinometer'. Horizontal elevation was measured at eight equidistant compass bearings. Aspect was measured using a compass with reference to magnetic north. Slope, Aspect and horizontal information was included within the analysis as raw scores. Additional attributes scored at each site included altitude, and logging, grazing and fire disturbances. The disturbance categories were scored based on a subjective severity score of one to five, with one being no visible signs of disturbance and five being obvious and severe disturbance. For instance a score of five for logging would include wholesale removal of all overstorey species, similarly a score of five for fire would include an intense and complete crown fire, and five for grazing would include heavy stocking rates within the past twelve months. The subjectivity of these disturbance scores has been reduced by recording by the same observer. Scores are subjective but considered appropriate for the level of analysis. The Braun-Blanquet score for *Callitris glaucophylla* and *Callitris endlicheri* was also used as a variable for each site. Multiple forward stepwise linear regressions of variables were carried out on the variance in the number of species per plot (species density).

Results

A summary of attributes of each dataset used for the analysis are presented (Table 1). No *Callitris* stands scored a foliage cover of more than 75% (score of 6). Overall no significant difference was found between any *Callitris* cover/abundance

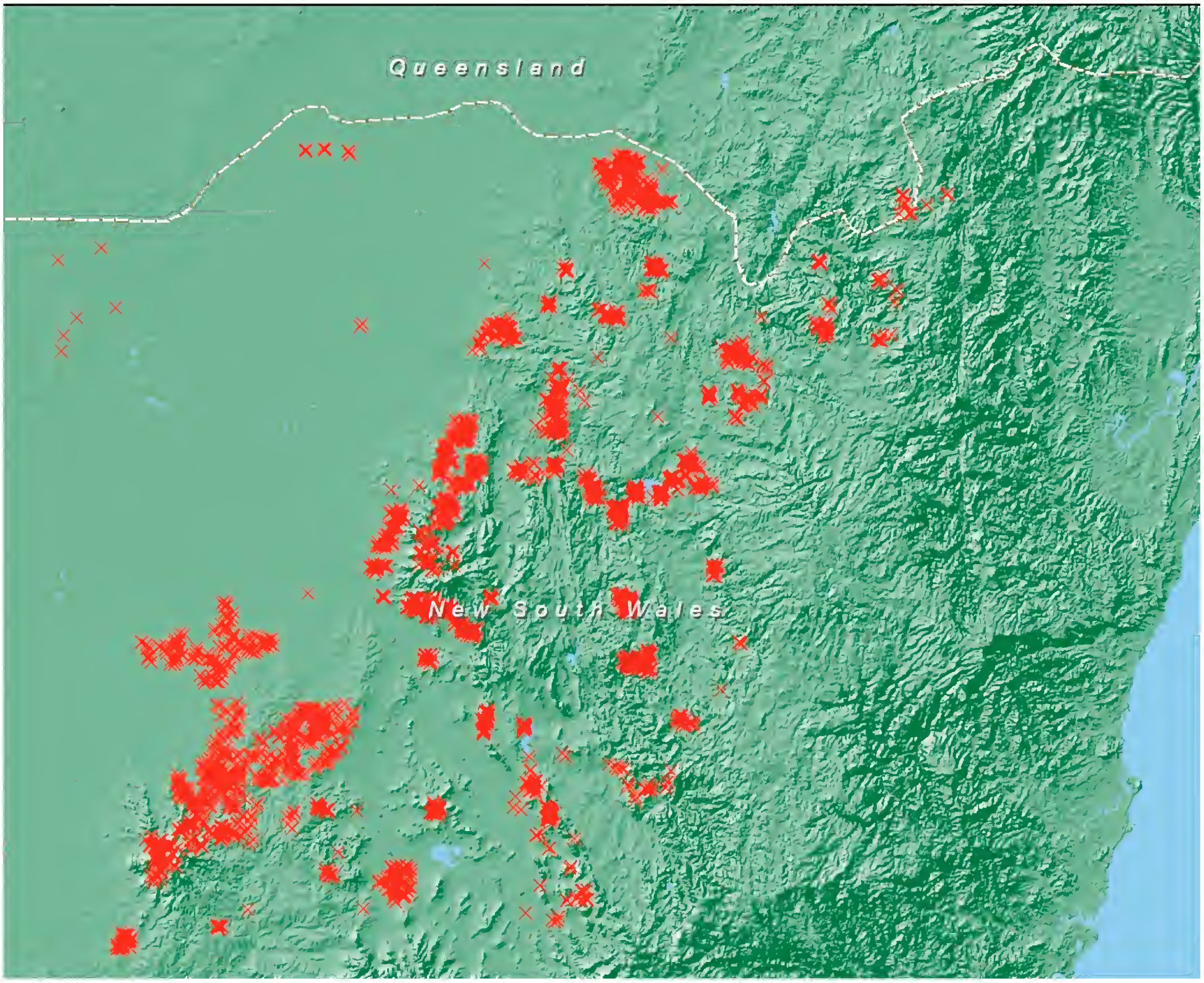

Figure 1: Location of sites within the study area.

Table 1. Selected attributes of the data used for the analysis

Selected Attributes	***Callitris glaucophylla***	***Callitris endlicheri***
Number of plots (sites)	809	542
Altitude range	50–1030 m	100–1210 m
Total number of taxa recorded	1032	995
Minimum number of taxa per plot	5	8
Maximum number of taxa per plot	84	93
Average number of taxa per plot	31	33
Percentage of plots with more than 30 taxa	45%	49%
Number of plots with less than 10 taxa	6	1
Number of Plots with *Callitris* cover/abundance score 1	182	146
Number of Plots with *Callitris* cover/abundance score 2	32	41
Number of Plots with *Callitris* cover/abundance score 3	315	150
Number of Plots with *Callitris* cover/abundance score 4	315	185
Number of Plots with *Callitris* cover/abundance score 5	14	30
Number of Plots with *Callitris* cover/abundance score 6	0	0

Table 2. Explanation of variation in species density (number of species per site) based on forward stepwise multiple linear regression in sites containing a) *Callitris glaucophylla* (*n* = 809). Overall variance explained 10% (p<0.00000) and **b)** *Callitris endlicheri* (*n* = 542). Overall variance explained 23% (p<0.00000). Only significant variables are presented.

a)

Variable	**Correlation**	***r^2 change***	***P***
1. Altitude	Positive	0.311	<0.00000
2. *Callitris glaucophylla* Cover/Abundance Score	Positive	0.093	0.00514
3. Fire	Negative	0.076	0.00235

b)

Variable	**Correlation**	***r^2 change***	***P***
1. Altitude	Positive	0.471	0.00000
2. Grazing	Positive	0.111	0.00610
3. *Callitris endlicheri* Cover/ Abundance Score	Not significant		

score and species density (number of taxa per plot) from either dataset (Fig 2a, b). Regressions on both *Callitris* datasets showed altitude as the most significant predictor of the variance in species density (number of taxa per plot), far in excess of any other variable analysed (Table 2a,b). The data showed a small positive effect of increased species density with increasing *Callitris glaucophylla* cover/abundance, and a reduced species density with increase in the severity of fire (Table 2a). In contrast no significant effect positive or negative was found on species density with an increase in *Callitris endlicheri* cover/abundance (Table 2b). Fire severity did not have a significant effect on the species density in *Callitris endlicheri* sites though grazing had a small positive affect (Table 2b). Logging severity was not found to have a significant effect on species density within either dataset.

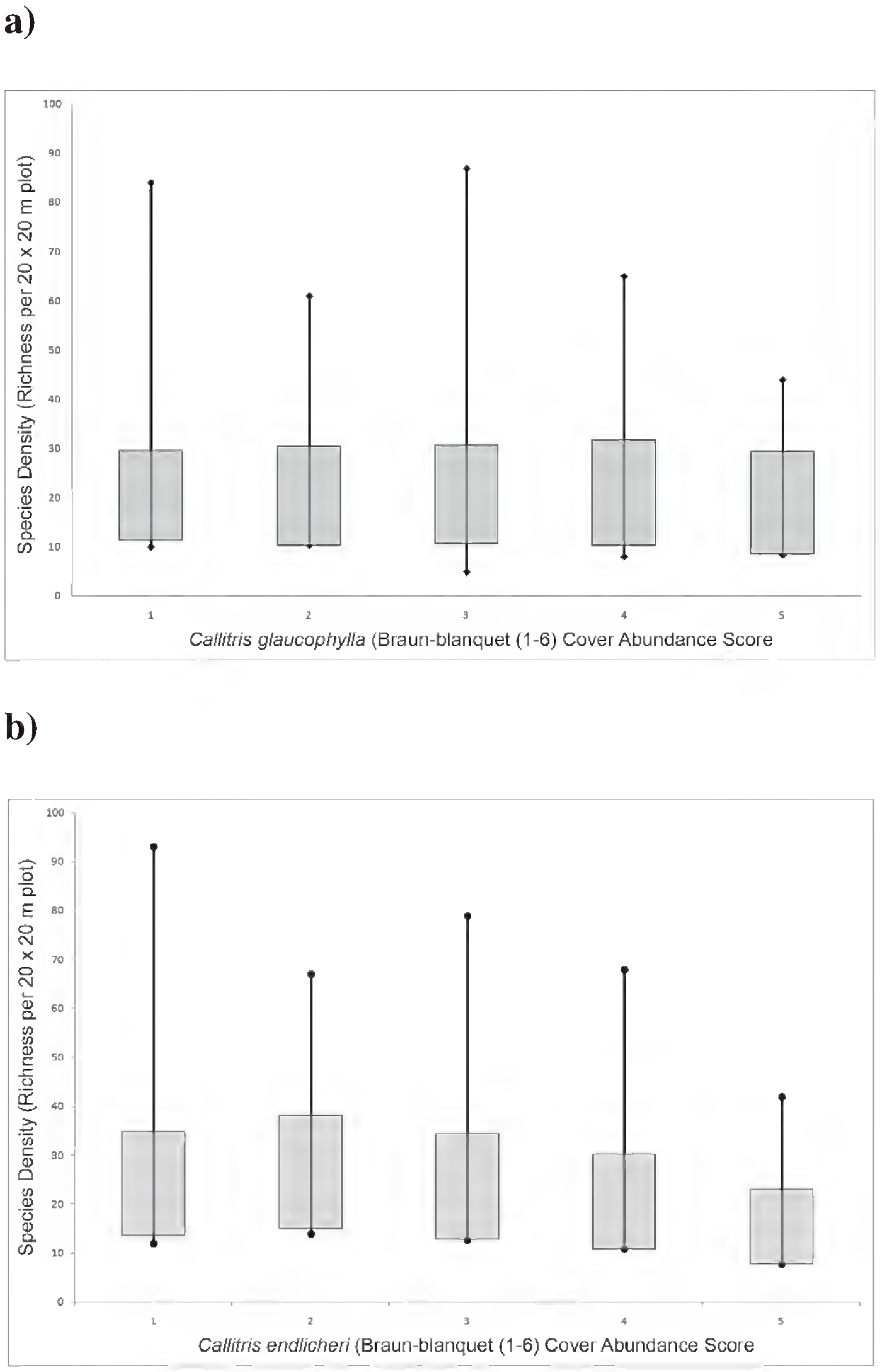

Fig 2. Variance in species density (number of species per site) for different cover abundance scores for **a)** *Callitris glaucophylla* and **b)** *Callitris endlicheri*. Maximum, minimum and standard deviation about the mean presented. Cover codes: 1 <5% few individuals; 2 <5% any number of individuals; 3 = 6–25%; 4 = 26–50%; 5 = 51–75%; 6 >75%.

Discussion

Multiple linear regressions of data from 1,347 sites across the New England, Nandewar and Brigalow Belt South Bioregions showed no negative effect of increasing *Callitris* cover on the number of species per site. This is in contrast to anecdotal reports from some researchers (Walker *et al.* 1972; Boland *et al.* 1984; Clayton-Green & Ashton 1990) but in agreement with others (Andrews 2003; Thompson & Eldridge 2005a). In their analyses Thompson and Eldridge (2005a) showed that rainfall, potentially soil texture and carbon, and past disturbance history were greater predictors of change in understorey species richness, density and cover. Their results parallel the strong correlation with altitude presented here. Altitude relates closely with rainfall in the region studied (Hunter 2003ab; Hunter 2005). Scale of investigation is of importance as it effects the importance of local and regional factors in regression studies, with local factors becoming less important as the scale of the investigation increases (Hunter 2002; Hunter 2005). Thus local topographic factors may be important at a smaller local scale but were found to be unimportant in this broad-scale investigation.

One point of difference to previous works is the increase in species density found with an increase in cover/abundance of *Callitris glaucophylla*. It is difficult to speculate on the reasons why richness may have increased in this instance without more on ground manipulative experimentation. It could potentially be that *Callitris glaucophylla* provides a cover that ameliorates the extremes of climate, allowing species to establish and/or persist. This was not seen with *Callitris endlicheri*. Fire was found to have a significant negative effect on species density in *Callitris glaucophylla* sites but not *Callitris endlicheri,* and conversely grazing had a positive effect on the species density of *Callitris endlicheri* sites but had no significant effect on *Callitris glaucophylla* sites. The reasons for these differences cannot be explained based on the information contained in this broad study and require further detailed investigation.

Species density (number of species per plot) is only one aspect of landscape management (Hunter 2005). However species density is a score that is commonly used as a surrogate for other diversity measures and widely used as an indicator for management decisions. The results of this brief study should not be used as a sole basis for management decisions; however they provide evidence against to the widespread belief that increased *Callitris* cover/abundance is necessarily detrimental to species density. Land managers with dense *Callitris* stands who are concerned about abundance of all species are faced with the challenge of whether to leave the stands as they are, or to use one or more of the various techniques available to thin them, such as burning or mechanical thinning (Thompson & Eldridge 2005b; Lunt *et al.* 2006). Different paths are likely to cause different long-term outcomes resulting in locations perhaps diverging markedly from each other. Which is the right choice or choices for biodiversity within the landscape is still open to question, and needs further manipulative research.

Acknowledgements

I wish to thank the many people who have assisted me over many years in the field.

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SHORT COMMUNICATION

Cool Temperate Rainforest in the Pilot Wilderness Area, Kosciuszko National Park, New South Wales: Distribution, composition and impact of the 2003 fires

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***Abstract*:** Although the distribution and composition of cool temperate rainforest in eastern Australia may be regarded as well documented, the recent discovery of cool temperate rainforest stands dominated by *Atherosperma moschatum* in the Pilot Wilderness area of Kosciuszko National Park shows that our knowledge is still incomplete. The additional discovery of 10 plant species previously unrecorded for the park including large specimens of *Elaeocarpus holopetalus* highlights the fact that although the flora and vegetation of the alpine and subalpine tracts of Kosciuszko National Park are relatively well studied, the remainder of the park is by comparison understudied and under sampled. Although not actively protected or managed, these cool temperate rainforest stands appear to have been little affected by the 2003 fires in the Australian Alps, with only 2 stands out of 25 showing any fire incursion. However, whether the direct effects of climate change or the indirect effects of human reaction to climate change poses the greatest threat to the continued existence of these stands is an open question. The aim of this short communication is to: a) examine the distribution and composition of these newly discovered stands of cool temperate rainforest and b) to briefly describe the impact of the 2003 fires on this restricted vegetation type.

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Introduction

Cool temperate rainforest in Australia is restricted to high rainfall, high elevation areas from south east Queensland, through New South Wales and Victoria and into Tasmania (Howard and Ashton, 1973, Australian Heritage Commission, 1987, Jarman et al., 1987, Cameron, 1987, Helman, 1987, Floyd, 1990a, Floyd, 1990b, Busby and Brown, 1994). Although the larger stands are dominated by *Nothofagus moorei* in the north and by *Nothofagus cunninghamii* to the south, stands in southern New South Wales lack *Nothofagus* and are dominated by *Atherosperma moschatum*, *Elaeocarpus holopetalus* and *Eucryphia moorei*, which occur in a '*Nothofagus* gap' between these northern and southern occurrences of the type (Floyd, 1990a, Floyd, 1990b, Keith and Sanders, 1990). Although cool temperate rainforest stands tend to be small, the focus on rainforest protection in the 1980s and subsequent mapping of rainforest types over the past 30 years has meant that their distribution has been regarded as well documented.

However, during vegetation mapping for the Southern Comprehensive Regional Assessment in the mid 1990s (see Gellie, 2005), one of us (MDD) identified discrete stands of vegetation with a distinct LANDSAT TM false colour signature in the vicinity of the upper Pinch River and along the Suggan Buggan Range in Kosciuszko National Park. At the time, it was thought that these were likely to be stands of *Eucalyptus fastigata* with a mesic understorey, typically consisting of species such as *Bedfordia arborescens* and *Olearia argophylla*, with occasional *Hedycarya angustifolia*. Such moist eucalypt forest types are known from the Leatherbarrel Creek area to the west of Dead Horse Gap (Wimbush and Costin, 1973). Although creek line stands of *Atherosperma moschatum* are known from the Geehi area in Kosciuszko (Martin, 1986, Good, 1992) and from the Bogong Peaks area, there are no herbarium or survey records of *Atherosperma moschatum* from the south eastern side of Kosciuszko National Park (Australian Virtual Herbarium). Subsequently, in revising and updating a comprehensive plant species list for Kosciuszko National Park (Doherty,

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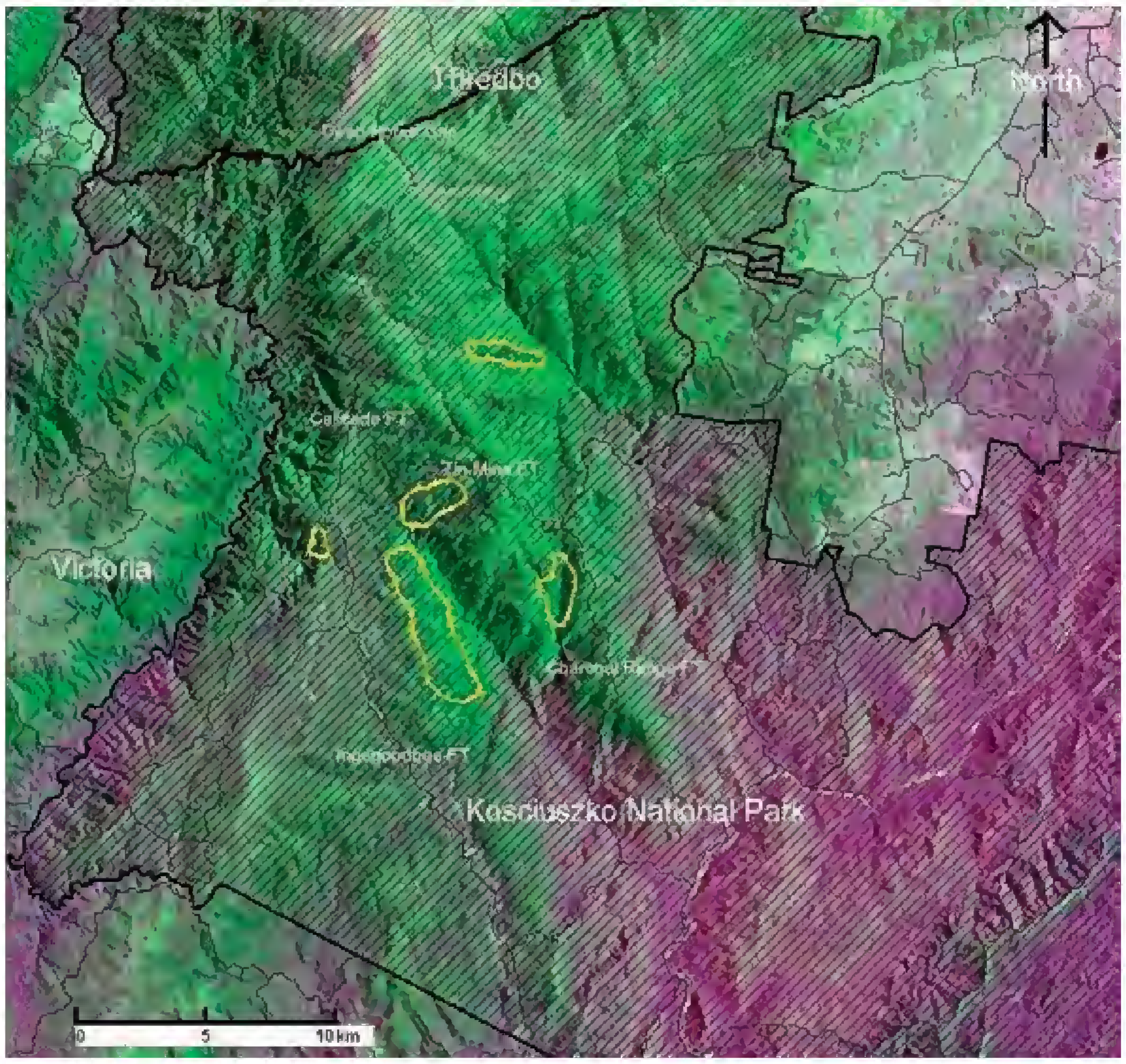

Fig. 1. Location of cool temperate rainforest in the Pilot Wilderness Area: areas within which discrete but non-contiguous stands occur are outlined in yellow; FT refers to Fire Trail. (False colour LANDSAT TM image, June 1999)

Wright and Duncan in prep.) the opportunity was taken to investigate these areas along the Suggan Buggan Range, particularly as independent field investigations by one of us (DC) after the 2003 wildfires had located creek line stands of *Atherosperma moschatum* in south eastern Kosciuszko National Park in the Charcoal Range area and in the upper Pinch River. Initial attempts by DC to access the largest stand proved unsuccessful, so a detailed inspection of the stands was undertaken in April 2010.

As the stands are small and restricted and embedded within more flammable eucalypt forest vegetation types, it is important to be able to accurately document their distribution, detail their composition and to assess any impact from the 2003 fires that may affect their viability. The aim of this short communication is to: a) examine the distribution and composition of newly discovered stands of cool temperate rainforest in Kosciuszko National Park and b) to briefly describe the impact of the 2003 fires on this restricted vegetation type.

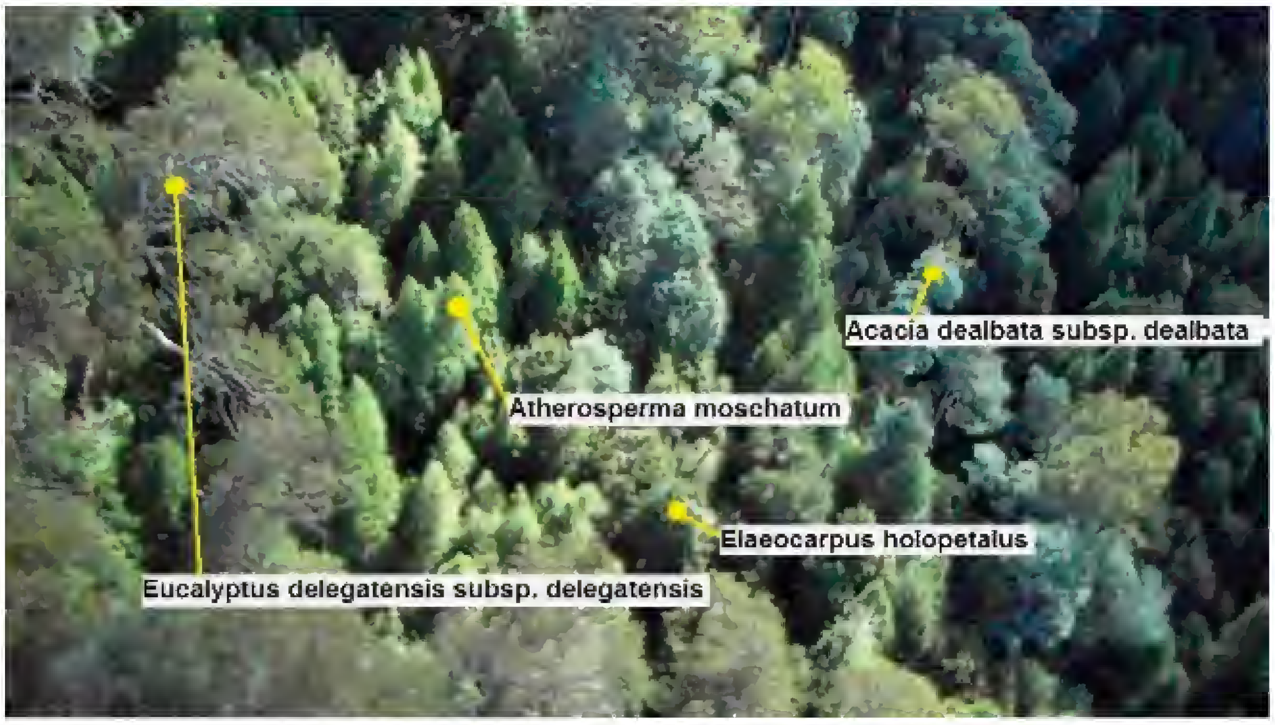
Acacia dealbata subsp. dealbata
Atherosperma moschatum
Elaeocarpus holopetalus
Eucalyptus delegatensis subsp. delegatensis

Fig. 2. Canopy types in the larger cool temperate rainforest stand. Oblique aerial image.

Methods

Initial inspection of false colour LANDSAT TM imagery (image: June 1999) in the target area of the upper Pinch River and Suggan Buggan Range in Kosciuszko National Park identified areas of high productivity vegetation, which characteristically show a bright green rather than dark green (moist vegetation types) or purple (dry vegetation types) canopy signature (Figure 1). Candidate stands of cool temperate rainforest vegetation were then identified within these areas using finer resolution imagery available via Google Earth (image: June 2004) and the altitude and grid reference, taken as the upper and lower extent of the stand, can be found in Appendix 1. A subsequent helicopter inspection was undertaken of the area and confirmed that most occurrences are narrow creek line stands of *Atherosperma moschatum*, generally only one or two trees in width. However, one stand was identified for further on-ground field investigation because it was significantly larger than the creek line stands and had crown types other than *Atherosperma moschatum* (Figure 2). The stand was sampled in April 2010, with a reconnaissance survey of its floristic composition and the completion of a permanent 20 m x 20 m reference plot. All vascular plant species encountered in the stand were recorded and on the reference plot, species cover was estimated using a modified Braun-Blanquet 1–6 cover scale and the DBH of a sub-sample of *Atherosperma moschatum* was measured. Mosses and liverworts were recorded opportunistically.

Results

The cool temperate rainforest stands in the Pilot Wilderness area are located in steep, narrow south or southeast facing drainage lines, ranging in elevation from 975–1405 m, with most stands occurring above 1200 m. The majority of stands in the area occur in the upper Pinch River catchment, bounded to the west by the Ingegoodbee and Cascade Fire Trails, to the north by the Tin Mine Fire Trail and to the east by the Charcoal Range Fire Trail, with one outlier stand in Tin Mine Creek and one stand in the upper Jacobs River catchment (Figure 1). All occurrences are on Mowamba Granodiorite, a unit which extends across a large area of southern Kosciuszko. The largest cool temperate rainforest stand is approximately 8 ha in area but as is typical for cool temperate rainforest, does not exhibit high plant species richness (18 species in

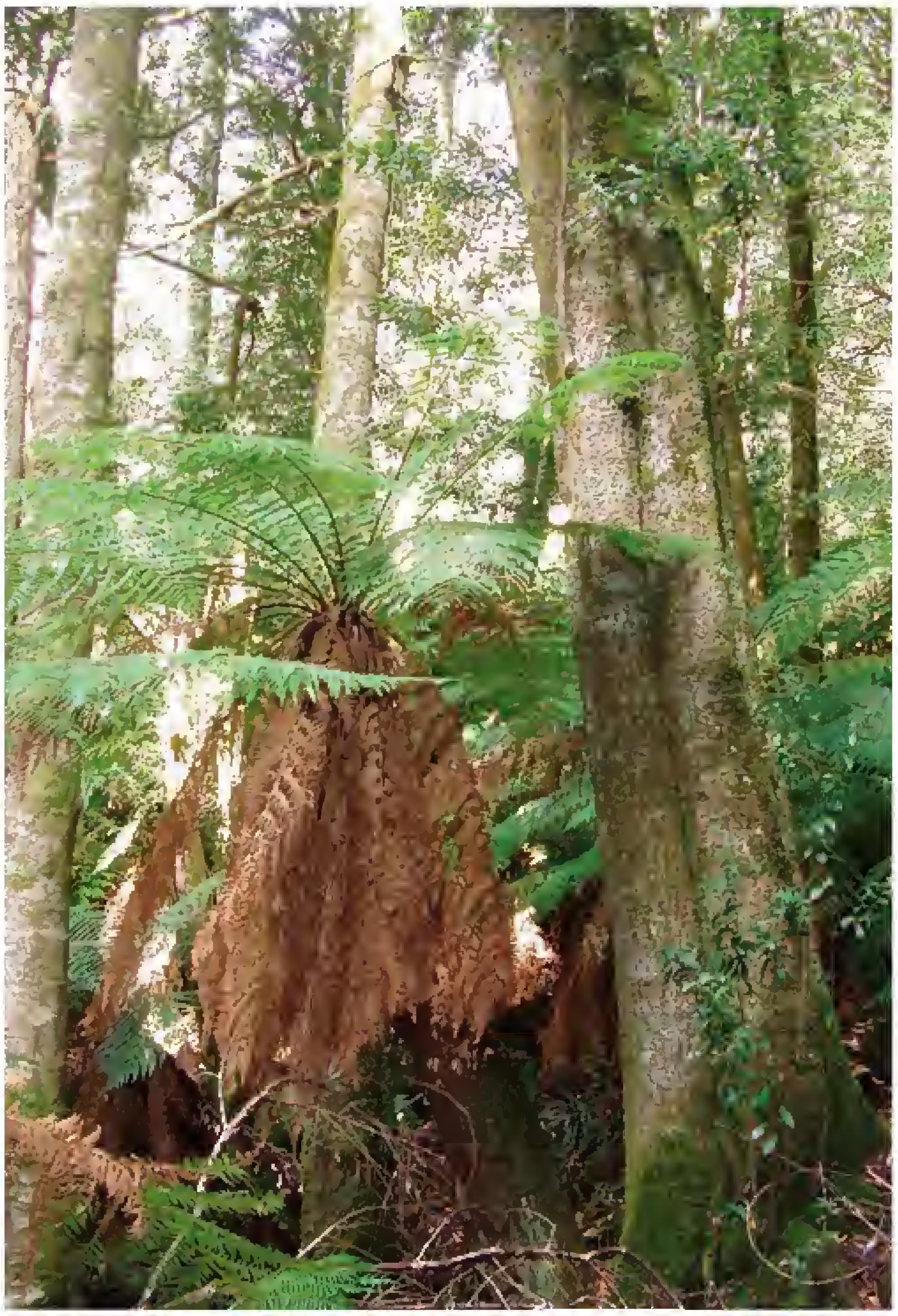

Fig. 3. Interior of stand with *Atherosperma moschatum* and *Dicksonia antarctica.*

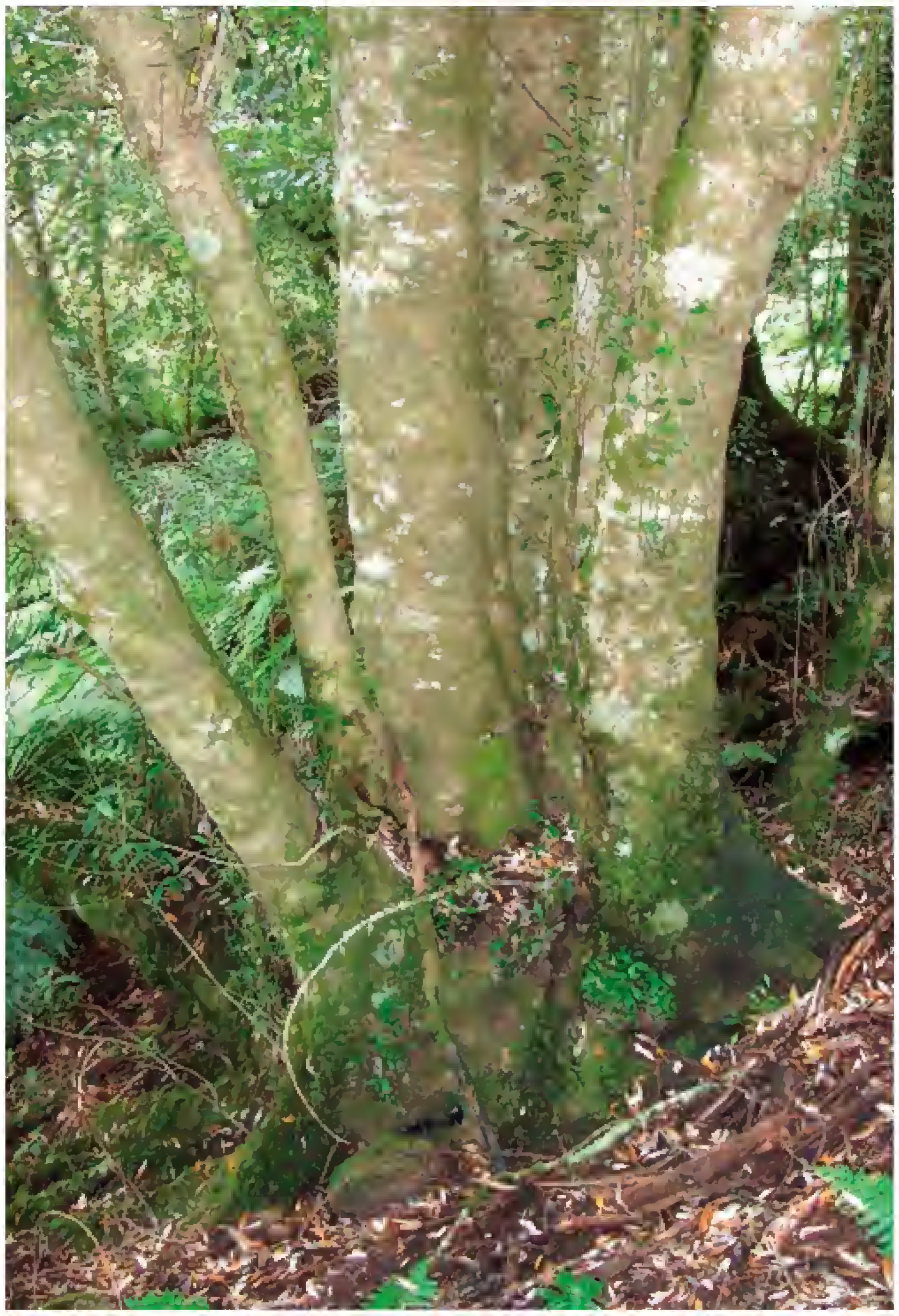

Fig. 4. Multi-stemmed specimen of *Atherosperma moschatum.*

total, Appendix 2). However, of the species so far recorded in the stand, ten are new records for Kosciuszko National Park. These species include *Blechnum patersonii*, *Blechnum chambersii*, *Uncinia tenella* and *Parsonsia brownii*, all of which are found in cool temperate rainforest stands further to the south and east. Of particular interest is the presence of large specimens of *Elaeocarpus holopetalus*, a species whose nearest known occurrences in New South Wales are 100 km to the east on the coastal escarpment and for which there were no known occurrences in the Park previously. Structurally, the canopy of the larger stand is dominated by *Atherosperma moschatum* to a height of 18 metres, with a diffuse sub canopy of *Elaeocarpus holopetalus*, a mid layer of *Dicksonia antarctica* and a ground layer of *Blechnum patersonii* and *Polystichum proliferum*.

The stand contained seedlings of both *Atherosperma moschatum* and *Elaeocarpus holopetalus* and also a range of saplings and adults, indicating ongoing recruitment. Diameter at breast height measurements for a sample of *Atherosperma moschatum* ranged from 15–46 cm (mean 27 cm) and heights from 12–18 m (mean 14 m). While most *Atherosperma moschatum* trees were single-stemmed (Figure 3) there were some multi-stemmed individuals (Figure 4) and while these may have been a result of coppicing from a past fire event, there was no evidence of fire or charcoal in the stand, despite the 2003 fires having burnt above and around the stand with high intensity. Johnston and Lacey (1983) note that multi-stemmed basal sprouting can occur in specimens of *Doryphora sassafras* and *Eucryphia moorei*, unassociated with disturbance by fire.

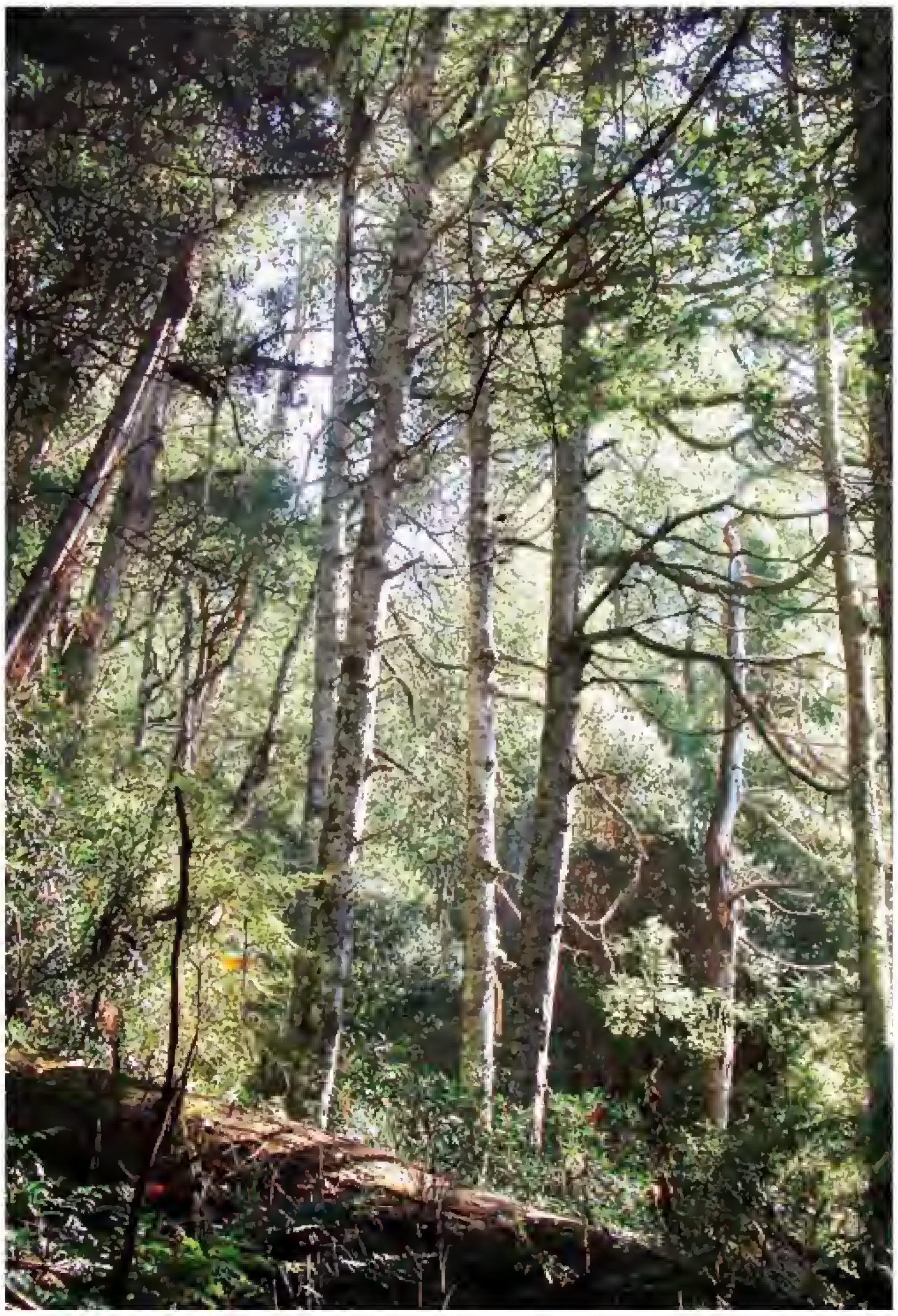

Fig. 5. *Acacia dealbata* subsp. *dealbata* on edge of cool temperate rainforest stand

Discussion

Floristic Affinities and Biogeography

The structure and floristic composition of the stand conforms to that of cool temperate rainforest (Cameron, 1987, Helman, 1987, Floyd, 1990a, Floyd, 1990b). Gellie (2005) notes cool temperate rainforest occurring in the Geehi catchment of Kosciuszko National Park although, the floristic composition of these stands is not detailed, other than their dominance by *Atherosperma moschatum*. Gellie describes this unit as VG 172: Kosciuszko Western Escarpment Cool Temperate Rainforest, which is a western outlier of VG 164: Coastal Escarpment Cool Temperate Rainforest, but does not contain *Eucryphia moorei* and up until the current survey, was also presumed not to contain *Elaeocarpus holopetalus*. Although Gellie estimated that 105 ha of VG 172 occurred in Kosciuszko National Park, the stands in the Pilot area and the Bogong Peaks area were not included in this estimation. However, as the recently discovered stands are generally confined to creek lines and the largest stand in the Pilot area is only 8 hectares, a total area of 120–150 ha may now be a reasonable estimation for the extent of this vegetation type in the Park. Communities with a co-dominance of *Atherosperma moschatum* and *Elaeocarpus holopetalus* occur further to the east and north on the high parts of the coastal escarpment in Mount Imlay National Park, Nalbaugh National Park and South East Forests National Park (Community 3, *Elaeocarpus holopetalus / Dicksonia antarctica* Closed-forest (Keith and Sanders, 1990); Suballiance 56: *Elaeocarpus holopetalus-Atherosperma* of (Floyd, 1990a, Floyd, 1990b)). Keith and Sanders note that *Eucryphia moorei* begins to occur as a dominant or co-dominant with *Atherosperma moschatum* and *Elaeocarpus holopetalus* on the coastal escarpment on more fertile soils. Areas such as Monga National Park contain communities where all three species co-occur (Floyd, 1990a, Floyd, 1990b).

The Pilot stands are transitional in relation to the overall composition of cool temperate rainforest in SE NSW. While *Atherosperma moschatum* occurs much further to the west in Victoria, *Elaeocarpus holopetalus* has its western limit only slightly further west of Kosciuszko on the Nunniong Plateau in Victoria (Peel, 1999). As *Elaeocarpus holopetalus* drops out, *Atherosperma moschatum* persists in very small isolated stands, including as isolated individuals in creek lines in western Kosciuszko National Park and at high altitudes on Mt. Buffalo in Victoria (Doherty, 2008). It has been noted that *Atherosperma moschatum* has greater long term drought resistance as compared to *Nothofagus cunninghamii* (Sommerville and Read, 2008) and this capacity may permit a greater westerly range than other cool temperate rainforest species in SE Australia. Although Olesen (1997) suggests that in mixed stands on the Errinundra Plateau *Atherosperma moschatum* will eventually outcompete *Elaeocarpus holopetalus* due to the former's greater shade tolerance, evidence from the Pilot stands suggest that *Elaeocarpus holopetalus* is able to recruit in these mixed stands. The larger stand in the Pilot area is floristically similar in composition to stands of Cool Temperate Rainforest found to the south-east on the Errinundra Plateau in Victoria (Cameron, 1987, Peel, 1999) where *Atherosperma moschatum* and *Elaeocarpus holopetalus* are co-dominant. The Pilot stand is essentially the same as Community 7, Sub-community 1, Cool Temperate Rainforest of Forbes et al. (1982), which is also equivalent to EVC 31: Cool Temperate Rainforest of Woodgate et al. (1994). However, at 1000–1400 m elevation, the Kosciuszko stands are generally higher than the stands found on the Errinundra Plateau, which occur between 800–900 m elevation.

Recent vegetation survey work in the Bogong Peaks area in tributaries of the Goobaragandra River (N. Taws, pers. comm.) has also detailed previously known occurrences of *Atherosperma moschatum* in the northern parts of Kosciuszko National Park and has found additional previously unrecorded species for the park including the ferns *Grammitis billardieri* and *Todea barbara*, as well as locating new populations of the ferns *Gleichenia dicarpa* and *Gleichenia microphylla*. These latter species have only been recorded sporadically in the park, often adjacent to stands of *Atherosperma moschatum*. Stands in the Geehi area also contain disjunct populations of ferns, in the latter case *Gleichenia microphylla* and *Sticherus urceolatus* (J. Miles pers. comm.).

The disjunct nature of the populations of both the stands and the species that occur within them reflect the fact that the stands are climatic relics, occurring only in small climatically suitable refuges where the species have been able to persist as Australia has moved northward since the breakup of Gondwana (Hill, 1994). For example, while *Nothofagus cunninghamii* is not present north of the Central Highlands of Victoria (Howard and Ashton, 1973), a stump of *Nothofagus*, probably *Nothofagus cunninghamii,* has been recorded from beneath a blockstream in the Toolong Range in Kosciuszko National Park, dated at approximately 35,000 years BP (Caine and Jennings, 1968).

Fire Ecology and Fire Management:

As is the case with mixed forests in the Victorian Central Highlands (Simkin and Baker, 2008), infrequent high intensity crown fires are the major disturbance agent in the *Eucalyptus delegatensis* subsp. *delegatensis* forest surrounding the cool temperate rainforest stands. Fire penetrating into the rainforest stands from surrounding eucalypt forest is therefore a major factor influencing the persistence of the rainforest stands. All stands of cool temperate rainforest identified in the study area occur in gullies or steep valleys with easterly aspects, providing protection from wildfires which generally run from the west or north west. There are no recorded fires in or immediately adjacent to the largest stand (in the upper Pinch River gorge) since at least 1939 (Office of Environment and Heritage GIS) and despite high intensity fire in 2003 in the surrounding eucalypt forest, only 2 of the 25 rainforest stands identified (Stand 2 and Stand 9) showed any evidence of fire penetration. The presence of *Acacia dealbata* subsp. *dealbata* adjacent to cool temperate rainforest stands is thought to indicate previous fire incursions (Peel, 1999) and this species was noted around the edges of the Pilot cool temperate rainforest stands (Figure 5). In particular, the presence of large specimens of *Acacia dealbata* subsp. *dealbata* and the absence of *Atherosperma moschatum* between Stand 9 and Stand 10 (Figure 5) may indicate a previous fire incursion that broke the connectivity of what may have been a continuous stand. However, the diameter at breast height of individuals of both *Acacia dealbata* subsp. *dealbata* and *Eucalyptus delegatensis* subsp. *delegatensis* on the margin of the largest stand (>50 cm and 1.9 m diameter respectively) suggests that it may be a century or more since a wildfire resulted in canopy death in or around the stand. Additionally, young individuals of *Atherosperma moschatum* downstream of Stand 10 indicate ongoing stand expansion in suitable areas.

Given the lack of fire penetration into the stands under even extreme conditions, this vegetation type appears to burn very infrequently and even when fire does penetrate the edges of a stand, there is still some ability for recovery. Some individuals of *Atherosperma moschatum* in the Geehi Valley in Kosciusko National Park were noted to resprout after the 2003 fires (D. Woods, pers. comm.), but there was also some tree death. Such attrition at the edges of stands may reduce their extent over the longer term, but the time between fire events will be critical, as a long interval between fire events may allow full recovery. Although generally classed as 'fire sensitive' communities, there is clearly some resilience in cool temperate rainforest communities in relation to particular fire events, as evidenced by the response of large individuals of *Nothofagus cunninghamii* which were observed resprouting after the 2009 Victorian fires (Lindenmayer et al., 2010). In Tasmania, *Atherosperma moschatum* can resprout vigorously from rootstocks after logging (Hickey and Wilkinson, 1999), 1999) and there is evidence of resprouting and seedlings after wildfire (Hickey, 1994), but Hill and Read (1984) noted almost 100% mortality of *Atherosperma moschatum* in a low to moderate intensity surface fire in western Tasmania.

Conclusion

As this work was undertaken as an initial reconnaissance, a systematic study of the occurrence and size of the stands in Kosciuszko National Park would be a logical next step, concentrating on the five largest stands initially (Appendix 1). Although considerable research and monitoring has been undertaken in the alpine and subalpine tracts of the Kosciuszko National Park over many years (McLuckie and Petrie, 1927, Costin, 1957, Costin et al., 1969, Wimbush and Costin, 1979a, Wimbush and Costin, 1979b, Wimbush and Costin, 1979c, Clarke and Martin, 1999, McDougall and Walsh, 2002, Walsh and McDougall, 2004, McDougall and Wright, 2004, Bear et al., 2006) few published studies exist outside these areas, exceptions being the lower Snowy River (Pulsford et al., 1993) and Byadbo areas (Clayton-Greene and Wimbush, 1988).

Given the large size (690,425 ha) and altitudinal variation (200–2228 m a.s.l.) of Kosciuszko National Park and the paucity of detailed survey work outside the alpine and subalpine tracts, the potential for unrecorded plant species and communities still remains significant, as evidenced by the number of new species records obtained for Kosciuszko National park during this survey.

The lack of fire penetration into the stands from the 2003 fires is encouraging and indicates a high degree of topographic protection from fire. However, as these relictual Kosciuszko cool temperate rainforest stands are small in overall extent and are often found as ribbon like threads only two or three trees wide along creek lines, projected climate and fire regime changes may result in an increase in the frequency with which fire burns around or into the stands and drier conditions may allow a greater depth of fire penetration into the stands in extreme weather situations, as experienced in 2003, or in 2009 in Victoria.

In the longer term, research on stand dynamics would be beneficial, particularly as regards the effects of predicted future changes in temperature and rainfall in SE NSW (PMSEIC Independent Working Group, 2007), and the possibility of an increase in the frequency of high intensity fire in SE Australia (Williams et al., 2001, Cary, 2002, Hennessy et al., 2005, Lucas et al., 2007). Such climate driven changes to fire regimes (Gill, 1975) i.e. the cumulative impact of fire events over time, may accelerate the climate-fire interactions that have already caused the extent of the communities to contract over many thousands of years.

Whether the direct effects of a possible increase in unplanned fire frequency and intensity prove more a threat to the existence of the stands than proposed increases in planned fire frequency and extent, as humans respond to climate change, remains to be seen.

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Appendix 1: Grid references for Cool Temperate Rainforest stands in the Pilot Wilderness

CTRF Stand Number	Altitude	Easting	Northing	Comments
CTRF Stand 1 Upper	1405 m	617920	5948354	Outlying stand
CTRF Stand 1 Lower	1225 m	618720	5948193	Outlying stand
CTRF Stand 2 Upper	1260 m	613904	5942584	
CTRF Stand 2 Lower	1235 m	613824	5942427	
CTRF Stand 3 Upper	1285 m	614330	5942539	
CTRF Stand 3 Lower	1250 m	614253	5942358	
CTRF Stand 4 Upper	1240 m	614222	5942290	
CTRF Stand 4 Lower	1210 m	614129	5942094	
CTRF Stand 5 Upper	1265 m	615105	5942227	
CTRF Stand 5 Lower	1160 m	614713	5941610	
CTRF Stand 6 Upper	1320 m	613439	5939811	
CTRF Stand 6 Lower	1140 m	614209	5940343	
CTRF Stand 7 Upper	1305 m	614176	5939064	Fifth largest stand
CTRF Stand 7 Branch	1305 m	614205	5939148	Fifth largest stand
CTRF Stand 7 Lower	1265 m	614381	5939223	Fifth largest stand
CTRF Stand 8 Upper	1320 m	614226	5938789	
CTRF Stand 8 Lower	1300 m	614280	5938981	
CTRF Stand 9 Upper	1230 m	614565	5938083	Largest stand
CTRF Stand 9 Centre	1120 m	614835	5938188	Largest stand
CTRF Stand 9 Lower	1065 m	615012	5938185	Largest stand
CTRF Stand 10 Upper	1330 m	614247	5937829	
CTRF Stand 10 Lower	1300 m	614357	5937906	
CTRF Stand 11 Upper	1350 m	614369	5936588	Second largest stand
CTRF Stand 11 Centre	1320 m	614484	5936593	Second largest stand
CTRF Stand 11 Lower	1310 m	614560	5936661	Second largest stand
CTRF Stand 12 Upper	1180 m	615308	5936962	
CTRF Stand 12 Lower	1150 m	615459	5936963	
CTRF Stand 13 Upper	1295 m	615548	5935926	Fourth largest stand
CTRF Stand 13 Lower	1250 m	615707	5935821	Fourth largest stand
CTRF Stand 14 Upper	1330 m	615079	5935361	
CTRF Stand 14 Lower	1275 m	615227	5935190	
CTRF Stand 15 Upper	1280 m	615180	5935095	Third largest stand
CTRF Stand 15 Branch	1270 m	615411	5935292	Third largest stand
CTRF Stand 15 Lower	1210 m	615723	5935097	Third largest stand
CTRF Stand 16 Upper	1245 m	615218	5934696	
CTRF Stand 16 Lower	1110 m	615707	5934363	
CTRF Stand 17 Upper	1280 m	614639	5934453	
CTRF Stand 17 Lower	1255 m	614611	5934319	
CTRF Stand 18 Upper	1040 m	616014	5934409	
CTRF Stand 18 Branch	1040 m	616130	5934671	
CTRF Stand 18 Lower	975 m	616359	5934503	
CTRF Stand 19 Upper	1340 m	619099	5939770	
CTRF Stand 19 Branch	1355 m	619061	5939556	
CTRF Stand 19 Lower	1045 m	619974	5938570	
CTRF Stand 20 Upper	1330 m	620308	5940736	
CTRF Stand 20 Lower	1065 m	620216	5938792	
CTRF Stand 21 Upper	1345 m	615066	5942734	
CTRF Stand 21 Lower	1255 m	615213	5942245	
CTRF Stand 22 Upper	1385 m	615812	5942986	
CTRF Stand 22 Lower	1350 m	615768	5942684	
CTRF Stand 23 Upper	1100 m	613929	5941462	
CTRF Stand 23 Lower	1095 m	613999	5941506	
CTRF Stand 24 Upper	1235 m	614577	5941908	
CTRF Stand 24 Lower	1065 m	614274	5941480	
CTRF Stand 25 Upper	1255 m	610410	5940926	
CTRF Stand 25 Lower	1000 m	610214	5940153	

Appendix 2: Plant species recorded in and immediately adjacent to the larger Cool Temperate Rainforest stand in the Pilot Wilderness Area

Species recorded in the reference plot in the core of the stand

Liverworts:

Hymenophyton flabellatum: **new record for Kosciuszko NP**

Mosses:

Cyathophorum bulbosum: **new record for Kosciuszko NP**
Hypnodendron vitiense: **new record for Kosciuszko NP**

Ferns:

Asplenium bulbiferum: **new record for Kosciuszko NP**
Asplenium flabellifolium
Blechnum chambersii: **new record for Kosciuszko NP**
Blechnum patersonii: **new record for Kosciuszko NP**
Dicksonia antarctica
Histiopteris incisa: **new record for Kosciuszko NP**
Polystichum proliferum

Herbs:

Viola hederacea hederacea

Shrubs:

Coprosma quadrifida
Pittosporum bicolor
Tasmannia lanceolata

Trees:

Atherosperma moschatum
Elaeocarpus holopetalus: **new record for Kosciuszko NP**

Climbers:

Parsonsia brownii: **new record for Kosciuszko NP**

Sedges:

Uncinia tenella: **new record for Kosciuszko NP**

Common species immediately adjacent to the stand

Acacia dealbata subsp. *dealbata* (Silver Wattle)
Bedfordia arborescens (Blanket Bush)
Cassinia aculeata (Dolly Bush)
Eucalyptus delegatensis subsp. *delegatensis* (Alpine Ash)
Eucalyptus fastigata (Brown Barrel)
Olearia argophylla (Musk Daisy Bush)
Poa sieberiana subsp. *sieberiana* (Snow Grass)
Urtica incisa (Stinging Nettle)

SHORT COMMUNICATION

Status of the Vulnerable shrub *Astrotricha crassifolia* (Araliaceae) in Brisbane Water National Park, NSW: an update

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Abstract: A resurvey (previously surveyed 2003–04) of the northern metapopulation of the listed Vulnerable shrub *Astrotricha crassifolia* (family Araliaceae) near Gosford, New South Wales, revealed six additional small subpopulations nearby, bringing to nine the total number, all in Brisbane Water National Park. While the stem count of the previously measured sites remained largely the same, the discovery of further subpopulations has increased the total known stem number to 1211 stems, with an area of occupancy of only 385 m^2. The majority (nearly 80%) of these subpopulations are very small, directly adjacent to roads, and remain vulnerable to park management and maintenance practices.

Astrotricha crassifolia is surviving due to its successful rhizomatous growth, but may not be reproducing from seed. This paper recommends some changes to management to reduce potential threats.

Cunninghamia **(2011) 12(2): 129–136**

Introduction

Astrotricha crassifolia Blakely (family Araliaceae), an endemic New South Wales shrub (Benson & McDougall 1993), is listed as Vulnerable, both under the NSW *Threatened Species Conservation Act 1995* and the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*. It is only known to occur in two separate disjunct areas, a 'northern metapopulation' near Gosford, north of Sydney, and a 'southern metapopulation' near Sutherland, south of Sydney; in Gosford and Campbelltown local government areas respectively. The Gosford metapopulation occurs in Brisbane Water National Park (BWNP), near Warrah Trig between Pearl Beach and Patonga, in an area known as the Waratah Patch.

Astrotricha crassifolia is a narrow-leaved erect shrub to 2m (Pellow et al. 2009) with a conspicuous indumentum of stellate hairs on most parts of the plant except the upper surface of the mature leaf (M.J. Henwood & R.O. Makinson in Harden 2002). Plants appear to grow in light shade (Benson & McDougall 1993), quite densely, in a number of small, isolated patches. Flowers are insect-visited but pollination details are unknown. Although fruit are produced, there are no reports of the species producing viable seed; there has been no evidence of seed germination or seedling growth in over 20 years of intermittent observations (Bob Makinson, pers. comm. 17/11/ 2010). *Astrotricha crassifolia* responds to slashing and fire by resprouting from rhizomatous growth. The Waratah Patch was burnt during an extensive wildfire in 1990 with some sections again being burnt in 2001, 2002 (Beckers & Offord 2010).

The NSW Office of Environment and Heritage (OEH) has given high priority to surveying sites and documenting threat status of *Astrotricha crassifolia*. This study updates a 2003–04 census of three subpopulations in the Waratah Patch, Brisbane Water National Park (Beckers 2004), and reports on new subpopulations and ongoing threats.

Methods

The three previously surveyed subpopulations of *Astrotricha crassifolia* (*Acras1, Acras2* and *Acras3*) are in the Waratah Patch (lat 33° 32' S; long 151° 17'E), a ridgetop (altitude 190m) of deep sandy lateritic loam (Somersby soil landscape unit of Chapman & Murphy 1989), in Brisbane Water

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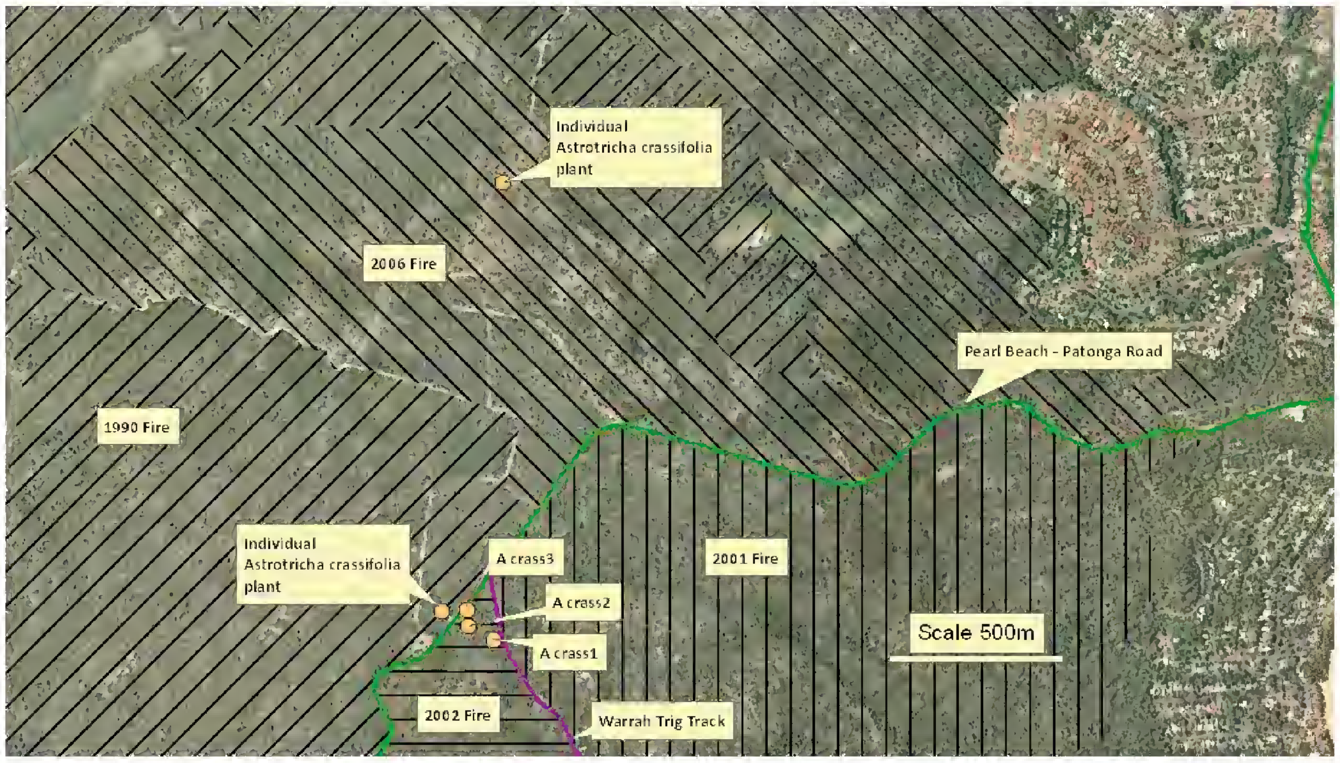
Individual Astrotricha crassifolia plant
2006 Fire
Pearl Beach - Patonga Road
1990 Fire
Individual Astrotricha crassifolia plant
A crass3
2001 Fire
A crass2
Scale 500m
A crass1
2002 Fire
Warrah Trig Track

Fig. 1. Aerial view of the Waratah Patch in Brisbane Water National Park showing subpopulations of *Astrotricha crassifolia* in 2004, and area of burns in 2001, 2002 and 2006.

National Park, between Pearl Beach and Patonga, about 80 km north of Sydney (Figure 1). Another two occurrences at the Waratah Patch, each consisting of a single plant, were not included in this study. In this survey, the term subpopulation has been used to describe discrete patches; however it is unclear whether or not the patches are of common clonal origin.

The three subpopulations (*Acras1*, *Acras2* and *Acras3*) surveyed in 2003–04, were relocated using a Garmin 12XL GPS (Figure 1). Plant species with rhizomatous root systems can be problematic to census because of interconnected ramets which may appear as individual plants or clumps above-ground. In some cases these may be extensive, and without excavation to determine whether they are separate plant clumps or joined, estimates of population size may be inaccurate (Payne 2002). In this case each stem, together with its immediately subtending portion of rhizome, constitutes a potential reproductive unit, and was therefore an appropriate measure of plant abundance. Each stem within a subpopulation was flagged, traced to the base, then measured and assigned to one of four height categories (shorter than 10cm, 10–30cm, 30–100cm and higher than 100 cm). An estimate of subpopulation size was determined by measuring the perimeter of the area bound by the flagged stems. Life history observations were recorded including presence of buds, fruit, and evidence of old or new flowers. Observed threats and disturbances were noted. A distance of approximately 2–5m beyond each patch was checked for any outlier plants and further subpopulations.

Following the census, an opportunistic field search was conducted of similar habitat adjacent to the Warrah Trig track to locate any further subpopulations of *Astrotricha crassifolia*, and any found were measured as previously described. Voucher specimens were collected and a list of associated plant species was compiled (Appendix 1). A detailed log of the hours taken to record each population was kept to provide estimates of time required to repeat the sampling in the future.

Table 1: Chronological burn and monitoring history of *Astrotricha crassifolia* subpopulations in the Waratah Patch, Brisbane Water National Park (to 2010 modified from Beckers 2004).

Date	*Acras* 1	*Acras* 2	*Acras* 3
1990	Burnt	Burnt	Burnt
1998			Disturbed by optic fibre cable laying
Nov 2002	Burnt	Burnt	Burnt
Dec 2003		468 stems Mean height 0.3 m	99 stems Mean height 0.3 m No plants in disturbed ground
Sept 2004	43 stems Some >1m tall Flowering		
Mar 2010	86 stems 71%> 1m tall No flowering material observed.	510 stems 111> 1m tall 50% had spent flowering material.	111 stems 47% >1m tall No flowering material observed.

Results

An update on status of 2003–2004 census

In 2003–04, around 600 *Astrotricha crassifolia* stems were recorded in the Waratah Patch (Table 1). In 2010, 707 stems were recorded for the same subpopulations, an increase of less than 20%. In September 2004, *Acras1* had 43 stems (with some over 1m tall); in 2010, 86 stems were recorded, 71% of which were in excess of 1m tall. In December 2003, *Acras2* had 468 stems (averaging 30cm tall); in 2010, 510 stems were found, an increase of less than 10%, 111 of which were greater than 1m tall. About 50 % of these stems had spent flowers. In December 2003, *Acras3* had about 99 stems (averaging 30cm tall), and in March 2010, 111 stems (an increase of less than 10%) of which only 47% reached a height of 1m.

Table 2. Size of *Astrotricha crassifolia* subpopulations, Brisbane Water National Park, in 2010 ordered by area of occupancy.

Population	Perimeter (m)	Population Diameter (m)	Area of occupancy (m^2)	No. of stems	Stems / m^2
Acras2	43	13.7	147	510	3.4
Acras3	35	11.1	97	111	1.1
Acras1	30	9.5	72	86	1.2
Acras7	21	6.7	35	290	8.2
Acras8	15	4.8	18	76	4.2
Acras9	9	2.9	6	39	6
Acras4	7	2.2	4	33	8.4
Acras5	7	2.2	4	34	8.7
Acras6	4	1.3	1	32	25.1

Field search and population size 2010

In winter, 2010, a further six small discrete subpopulations were located on the sandy lateritic loam within 5m of the Warrah Trig track. Some of the additional subpopulations were found on the opposite side of the Warrah Trig track to the existing known subpopulations (Figure 2). Those on the east of the track had been burned in 2001 as opposed to the western side of the track which was burned in 2002 (Figure 1).

The nine *Astrotricha crassifolia* subpopulations in the Waratah Patch had a total of 1211 stems(Table 2), an increase of about 50% on the number known in 2003–04. This represents an increase in the known size of the population, not an increase in the population size since 2003–4 as the plants are evidently long-lived and are likely to have been present in 2003–04, but not known. Most subpopulations are characterised by similar-sized plants suggesting widespread population recovery following a similar trigger event, probably the fires of 2001 and 2002. With the exception of *Acras4* (0%), most subpopulations had a high percentage of stems greater than 1m tall (Figure 3) and all, except for *Acras4* had more than 76% of plants greater than 30cm tall. There were very few subpopulations with plants in the 0–10cm height category. *Acras4* is subject to repeated slashing, and included 91% of plants less than 30cm tall.

The maximum perimeter length (43m) for *Acras2*, the largest subpopulation, equates to a patch diameter of 13.7m, suggesting an area of occupancy of 147m^2 and stem density of 3.4 stems/m^2 (Table 2). Using these approximate diameters, stem densities for all patches calculated to between 1 and 25 stems per m^2 (Table 2). The total area of occupancy of the *Astrotricha crassifolia* metapopulation in the Gosford LGA is 385m^2, this figure being an aggregate of all subpopulations. (Surveying the subpopulations took about 32 hours.)

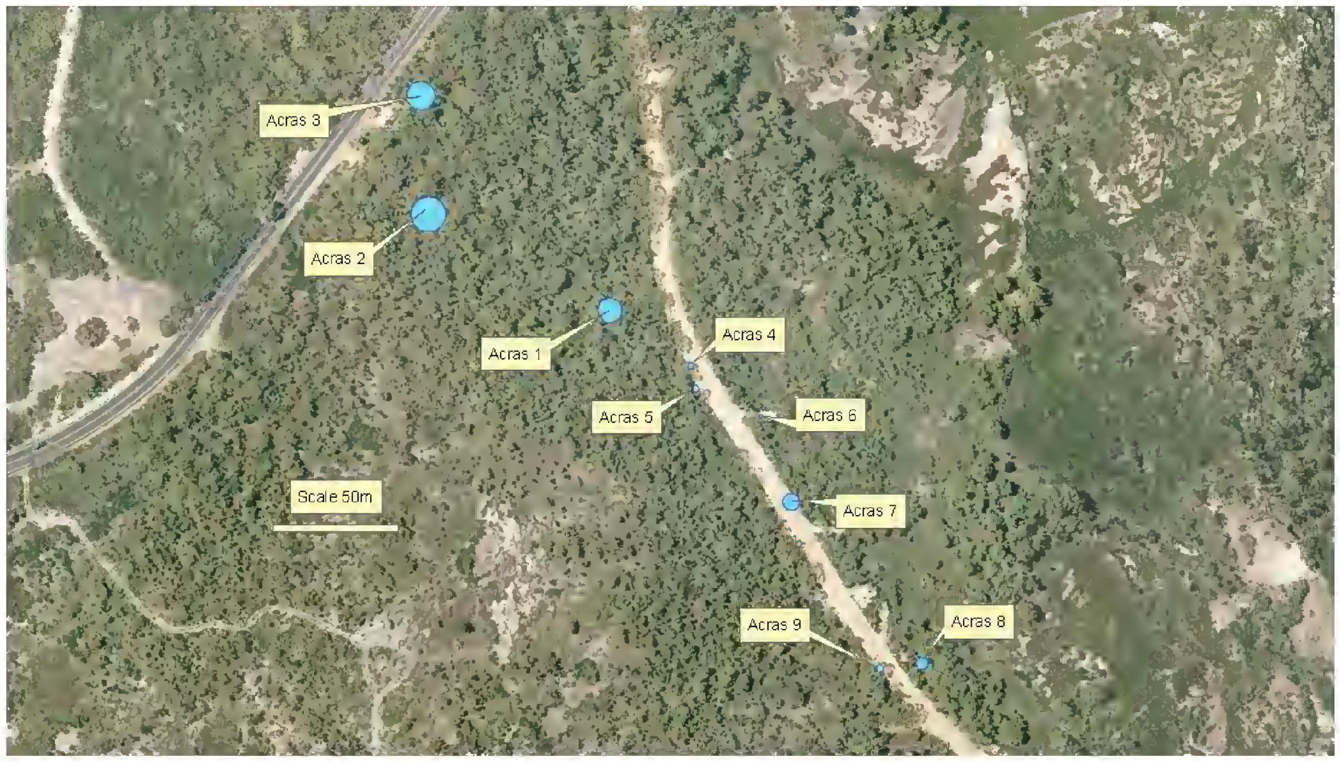

Fig. 2. Location of nine subpopulations of *Astrotricha crassifolia* recorded in 2010, Brisbane Water National Park.

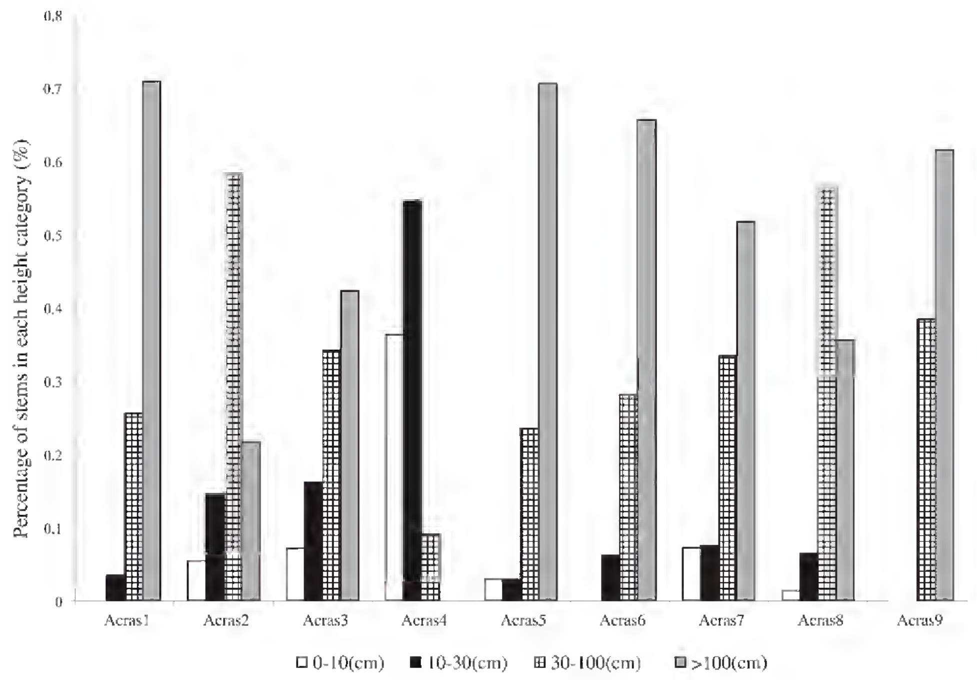

Fig. 3. Percentage of *Astrotricha crassifolia* stems in each height category in nine known subpopulations, Brisbane Water National Park, NSW.

Ecology

The vegetation community was generally low woodland to low open woodland dominated by trees of *Eucalyptus oblonga, Eucalyptus haemastoma, Corymbia gummifera.* The understorey at each subpopulation consistently included the Proteaceae shrubs *Banksia serrata, Conospermum longifolium*, *Grevillea buxifolia* subsp. *buxifolia*, *Isopogon anemonifolius, Lambertia formosa;* Fabaceae species included *Gompholobium grandiflorum*, *Phyllota phylicoides*, and Rutaceae species included *Boronia floribunda* and *Boronia ledifolia* (Appendix 1).

During late spring 2010, eight of the nine subpopulations flowered, except the slashed roadside population (*Acras4*) which was devoid of floral material. A native bee (identified by Anne Dollin (pers. comm. 2010) as a Reed Bee – Apidae *Exoneura* species) was observed on anthers of flowers in *Acras9* (Figure 4). Mass flowering of *Astrotricha crassifolia* appeared synchronised with *Conospermum longifolium* (Proteaceae) which shares a similar conspicuous paniculate inflorescence, and similar sized, white-coloured flowers to *Astrotricha crassifolia.* Stands of *Conospermum longifolium* were growing at similar heights to stands of *Astrotricha crassifolia* and were present in all of the nine subpopulations (Appendix 1).

All subpopulations (with the exception of slashed plants on the roadside) were observed flowering and producing fruit, and what appeared to be ripe fruit were observed on some plants (Figure 7). Later dissection of a small number of fruit revealed insect predation; no complete seed was observed and further research is required to determine whether viable seed is being produced.

The reproductive capacity of *Astrotricha crassifolia* may be influenced by vegetation density and light availability. Although the vegetation cover at each subpopulation was not measured, observations indicate that, consistent with Benson and McDougall (1993), *Astrotricha crassifolia* favours light shade. This study found that where the vegetation was particularly dense, the plant will extend robust branches more than 1m laterally through dense vegetation and leaf litter, in order to reach appropriate light intensity, often collapsing, forming an 'elbow' vertically towards the available light gap. However, where there was little competition for light and space the plant grew erect.

Resprouting from basal vegetative material, the recruitment strategy indicated in Benson & McDougall (1993), enabled *Astrotricha crassifolia* to recover from slashing for roadside maintenance. Very few stems were observed in the 0–10cm height category (Figure 3); what appeared to be seedlings in the leaf mulch were often found to be the vertical light-seeking shoots of collapsed branches.

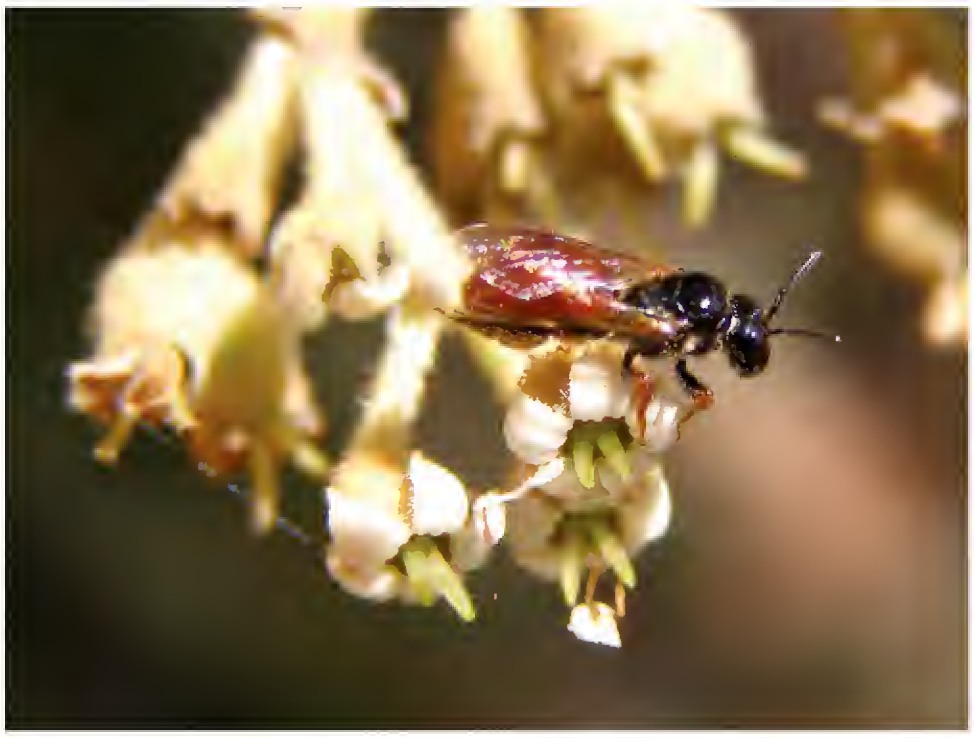

Fig. 4. Native bee (*Exoneura* sp.) on flowers of *Astrotricha crassifolia* (Photo D. Warman).

Fire

All subpopulations have been subjected to wildfire. In the 2004 inspection of the area burnt in 2002, Beckers (2004) found regenerating plants of *Astrotricha crassifolia* flowering. Unplanned fires in the area have affected all subpopulations; all have been burnt twice since 1990. The subpopulations on the east of the Waratah Patch are included within an area proposed to be burnt in the near future in a prescribed burn aimed at protecting the built assets in Pearl Beach (see Discussion regarding *Astrotricha crassifolia* post-fire response to this burn).

Human-induced threats and disturbance

Some subpopulations have been affected by a range of human-related impacts (Table 3) and 80% of the subpopulations are beside roads where issues include dust

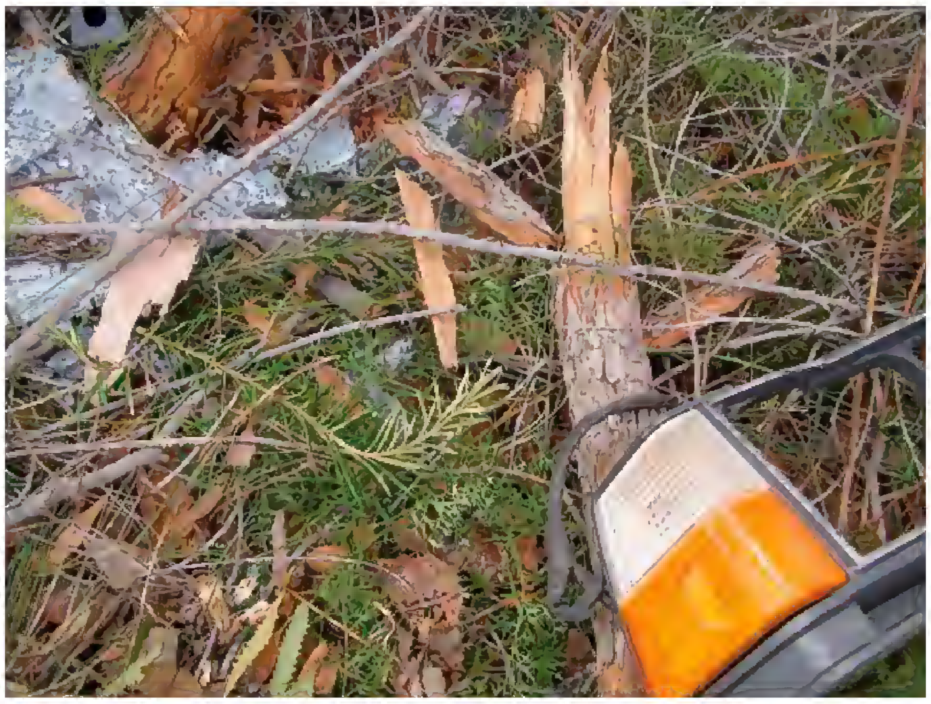

Fig. 5. *Astrotricha crassifolia* affected by vehicular damage along Warrah Trig Track. (Photo D. Warman).

from the unsealed Warrah Trig track, slashing, effects of runoff, and microclimatic changes due to sedimentation traps, edge effects and vehicular damage (Figure 5). The subpopulations directly adjacent to Warrah Trig track may have been fragmented during the original construction of the road, which may also have resulted in the loss of some plants. Illegal rubbish dumping occurs regularly in the area and during this survey 150 tyres were found dumped in the bushland adjacent to the road in very close proximity to occurrences of *Astrotricha crassifolia* (Figure 6).

The Waratah Patch is a very popular area to view Waratahs (*Telopea speciosissima*) and in peak Waratah flowering season, pedestrian tracks trample the vegetation. This has the potential to affect subpopulations *Acras1, Acras2* and *Acras3* which co-occur with Waratahs (Appendix 1).

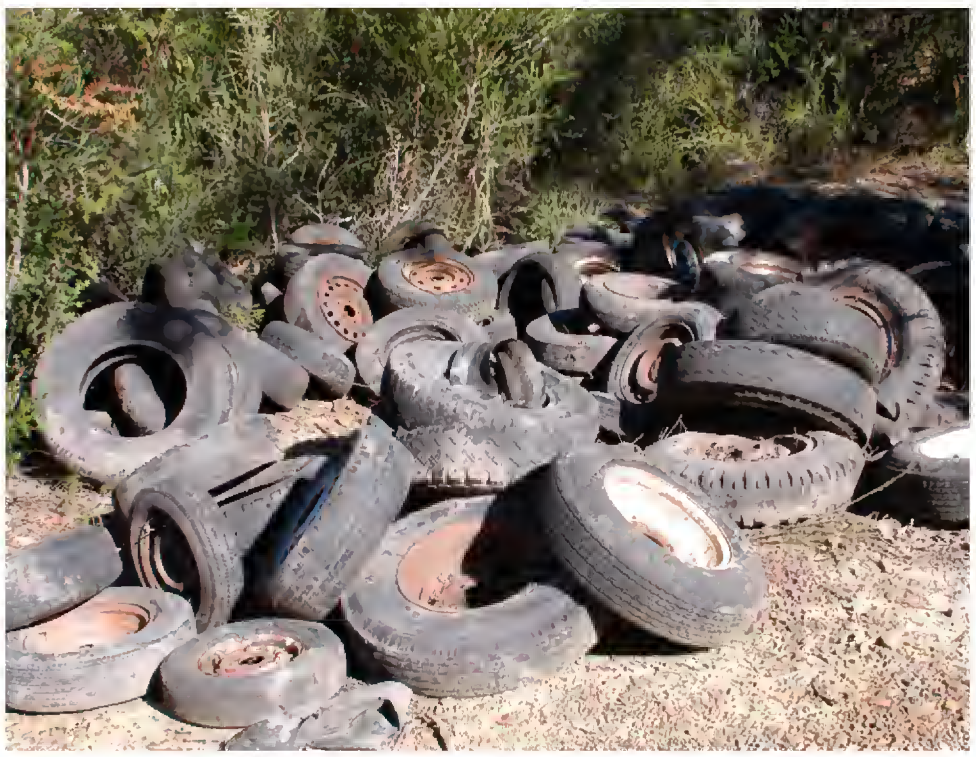

Fig. 6. Tyres dumped along the Warrah Trig track in very close proximity to roadside *Astrotricha crassifolia* subpopulations (Photo D. Beckers).

Discussion

The southern metapopulation of *Astrotricha crassifolia* (Campbelltown LGA) extends over a 5 km range in three habitat types – Riparian scrub, Gully forest and Low woodland; plants exhibit robust erect growth to 2m (Elizabeth Norris, pers. comm. May 2010). The northern subpopulations (Gosford LGA) of this study are restricted to a small number of disjunct clumps over a 1 km area of Low woodland-Low open woodland on a sandy ridgetop. Why the northern meta population is restricted to a single habitat is not known but the historic records influenced our search effort to target ridgetop habitats; in light of the southern population records, future targeted searches should not necessarily be confined to plateaus, but include slopes and riparian areas.

The nine subpopulations of the northern metapopulation of *Astrotricha crassifolia* were small clumps with a combined area of occupation of only 385 m^2; nearly 80% of those subpopulations are small, disjunct and adjacent to roads. With six subpopulations in close proximity to the Warrah Trig track, and the track potentially dissecting them, long-term conservation poses a major challenge for management. Management of park infrastructure (the reproductive potential of at least one small subpopulation was directly reduced by roadside slashing during this survey period) and other impacts associated with park use (e.g. dust from the unsealed road, vehicular damage, foot trampling and dumped waste) have the potential to further increase the vulnerability of these subpopulations and reduce their distribution. Slashing equipment and vehicle movement are vectors of weed seed and propagule dispersal, particularly grass species (the invasion of native plant communities by exotic perennial grasses is listed as a Key Threatening Process under the NSW *Threatened Species Conservation Act 1995*) and the

Table 3 : Human-induced disturbance events affecting individual *Astrotricha crassifolia* subpopulations (*Acras* 1–9), Brisbane Water National Park.

Disturbance event	*Astrotricha crassifolia* subpopulations								
	1	2	3	4	5	6	7	8	9
Potential fragmentation due to road alignment				•	•	•	•	•	•
Powerline easement and optical fibre maintenance			•						
Roadside maintenance, including slashing, sediment traps and road surface maintenance				•	•	•	•	•	•
Trampling by vehicle due to inappropriate access		•					•		
Trampling by recreational use – wildflower viewing	•	•							

exotic perennial grasses *Andropogon virginicus*, *Eragrostis curvula* and *Hyparrhenia hirta* already exist in comparable habitat in the park. The increased light radiation and moisture runoff along road edges may favour those weed species in competition with *Astrotricha crassifolia* plants.

To reduce their effect on some subpopulations, management practices could be altered. For example, clear identification of each roadside population may enable better planning of future infrastructure installation such as sedimentation traps. Machinery engaged for slashing and road-grading should incorporate hygiene practices in their preparation, prior to entering the track. Maintenance of existing sedimentation traps could be altered to consider reducing the effect on some subpopulations, while slashing in some areas may be avoided, or the slash height reduced or frequency reduced to allow flowering and fruit dispersal.

The NSW Office of Environment & Heritage website (OEH 2011 2) indicates inappropriate fire regimes are a threat to the species, though the NSW National Parks and Wildlife Fire Response Database (NSW NPWS 2002) indicates there are not enough data to inform an appropriate fire regime for the species. The 2004 Census and our current work have increased knowledge of the species response to fire. Park fire management needs to ensure an appropriate regime is maintained. Knowledge of this plant's response to fire is crucial given the significance of the Waratah Patch to local tourism (Beckers & Offord 2010) and to reduce competing priorities for Waratah and fuel management. As this paper goes to press we can document the plant's response to a hazard reduction burn (on 22/9/ 2011) on the eastern side of Warrah Trig track, in the vicinity of subpopulations (*Acras 6,7,8*). On 11/11/2011, no true seedlings were observed but *Astrotricha crassifolia* resprouts were emerging from the soil at the base of burnt stems (Figure 7).

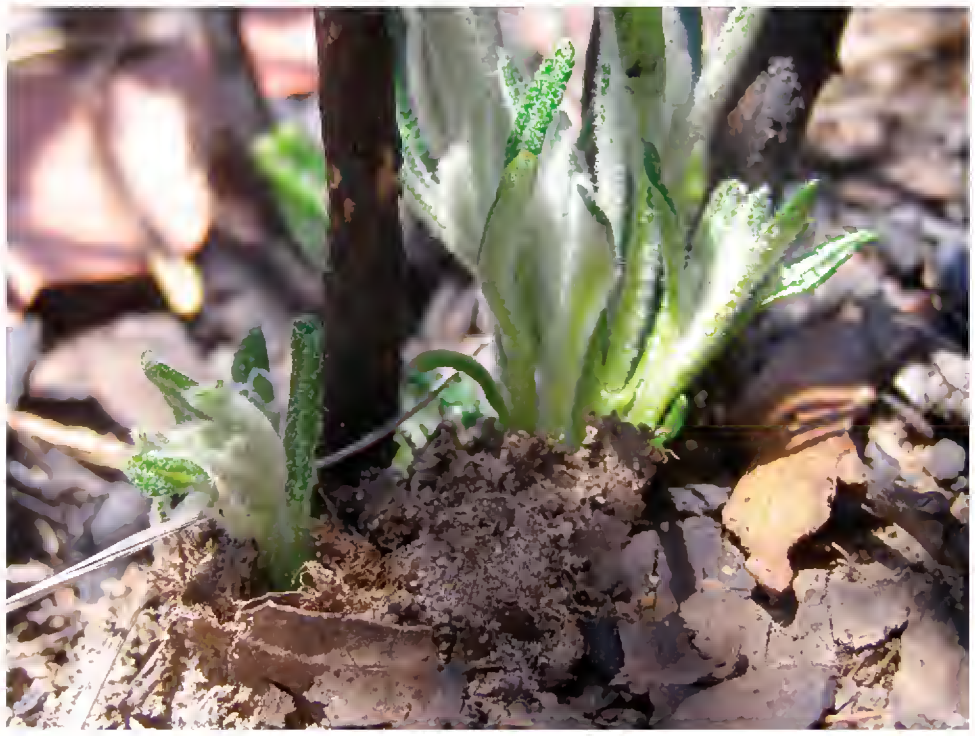

Fig. 7. Response to hazard reduction burn, *Astrotricha crassifolia* resprouting in November 2011, Brisbane Water National Park (Photo D. Warman).

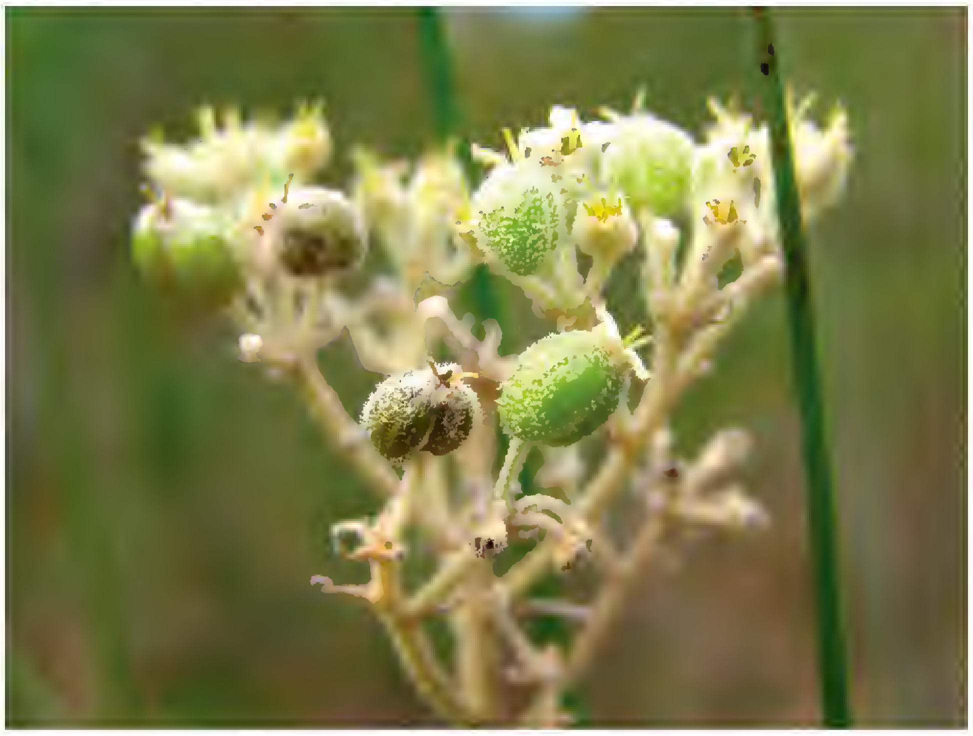

Fig. 8. Mature fruit of *Astrotricha crassifolia*, Brisbane Water National Park, 7 Dec 2010 (Photo D. Warman).

High frequency fire is considered a key threatening process to many plants, but Gross (2006) identifies that the literature on clonality, implies frequent fire may which favour clonal species. There may however be serious long-term implications in clonal plants with the loss of sexual reproduction e.g. clonality may induce a cascade of events leading to sexual extinction and finally complete genetic homogenization (Honnay et al. 2006), and susceptibility to pathogens and disease (Schmid 1994). However, despite the limitations, evidence of extraordinary ages of clones, e.g. *Eucalyptus recurva*, a mallee eucalypt may have survived up to 13,000 years without sexual reproduction (NPWS 2003), means that being a clone may provide an alternative survival strategy. While we cannot confirm clonality in *Astrotricha crassifolia*, its localised patches of stems and the absence of evidence of true seedlings warrants the question as to whether the subpopulations are in fact clones.

The state and national conservation significance of *Astrotricha crassifolia* suggests that further research is a priority. Using its habitat features from this survey, it would be appropriate to model the distribution of *Astrotricha crassifolia* to predict additional localities for targeted searching. Information on the southern metapopulation could be incorporated into future modelling and a comparison of population genetic structure of the two disjunct populations would be interesting.

Conclusion

The northern metapopulation of *Astrotricha crassifolia* remains vulnerable, in the short term, to human-induced impacts, and in the long term to a lack of understanding of its reproductive success and how fire affects its recruitment and survival. The following recommendations are made to reduce the threats:

- Limit infrastructure management activities alongside Patonga Road and Warrah Trig Road near *Astrotricha crassifolia* subpopulations during the spring flowering and summer fruiting period i.e. August to February;
- Incorporate hygiene practices into roadside management activities;
- Further identify the geographical limits of the northern metapopulation;
- Undertake post-fire monitoring of subpopulations to determine fire response, i.e. potential seed viability, germination, seedbank dormancy and longevity;
- Expansion of monitoring to include the other two subpopulations in BWNP (Figure 1);
- Ascertain the degree of clonality within and among *Astrotricha crassifolia* subpopulations using molecular techniques.

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Special thanks to Bob Makinson (Royal Botanic Gardens & Domain Trust) who provided useful insights into the taxonomy and ecology of the species and to Doug Benson (Royal Botanic Gardens & Domains Trust) for comments on the draft; their contributions are gratefully acknowledged. Also to Elizabeth Norris (Eco Logical Australia) who provided valuable information on the southern metapopulation. Thanks also to Barbara Wiecek and Andrew Orme (Royal Botanic Gardens & Domain Trust) who provided botanical information support through the highly useful Public Reference Collection, at The Gardens. Anne Dollin (Australian Native Bee Research Centre) kindly identified the native bee. Thanks also to Alan Henderson, Area Manager, Brisbane Water National Park (NSW National Parks & Wildlife Service, Gosford) for his ongoing support. Dr Anita Chalmers (University of Newcastle) made early comments on the presentation of results. Thanks also to an anonymous reviewer for worthwhile comments on the draft.

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Appendix 1.

Plant species recorded and voucher specimens collected from ***Astrotricha crassifolia*** subpopulations (Acras 1–9) along Warrah Trig Road, Brisbane Water National Park, during the 2010 census.

Species	Family	Observed present in
Acacia myrtifolia	Fabaceae	1–8
Acacia suaveolens	Fabaceae	1–3, 7–9
Actinotus minor	Apiaceae	1,2,4,5,9
Allocasuarina distyla	Casuarinaceae	9
Angophora hispida	Myrtaceae	6,7,8,9
Anisopogon avenaceous	Poaceae	1,2,3,5–9
Astrotricha crassifolia	Araliaceae	1–9
Baeckea diosmifolia	Myrtaceae	5,6
Banksia oblongifolia	Proteaceae	6,7,9
Banksia serrata	Proteaceae	1,2,3,5–9
Banksia spinulosa	Proteaceae	1,2,3,5
Billardiera scandens	Pittosporaceae	1,2,3,5
Boronia floribunda	Rutaceae	1,2,4,5,6
Boronia ledifolia	Rutaceae	7,8,9
Bossiaea heterophylla	Fabaceae	1–9
Bossiaea scolopendria	Fabaceae	4,5,6,7
Bossiaea stephensonii	Fabaceae	3
Caesia parviflora	Anthericaceae	1,4
Calytrix tetragona	Myrtaceae	3
Cassytha pubescens	Lauraceae	1,2,3,5,6
Chordifex dimorpha	Restionaceae	6
Comesperma ericinum	Polygalaceae	2,3
Conospermum longifolium subsp. *longifolium*	Proteaceae	1–9
Corymbia gummifera	Myrtaceae	1–9
Cyathochaeta diandra	Cyperaceae	1–9
Dampiera stricta	Goodeniaceae	1,2,3,5,6,8
Daviesia alata	Fabaceae	1,3
Dillwynia elegans	Fabaceae	4–9
Entolasia stricta	Poaceae	3–7
Epacris microphylla	Ericaceae	4
Eriostemon australasius	Rutaceae	8
Eucalyptus haemastoma	Myrtaceae	1–9
Eucalyptus oblonga	Myrtaceae	1,2,3,6,7,8
Gompholobium grandiflorum	Fabaceae	1–9
Gonocarpus teucrioides	Haloragaceae	7,8,9
Goodenia bellidifolia	Goodeniaceae	3,4,5,7,8,9
Grevillea buxifolia subsp. *buxifolia*	Proteaceae	1–9
Grevillea sericea	Proteaceae	1–4,6–9
Hakea dactyloides	Proteaceae	1,2,5,6,7
Hakea gibbosa	Proteaceae	4,5,9
Hibbertia cistiflora	Dilleniaceae	6,7,9
Hibbertia empetrifolia subsp. *empetrifolia*	Dilleniaceae	1–9
Hovea heterophylla	Fabaceae	2–5
Isopogon anenomifolius	Proteaceae	1–9
Lambertia formosa	Proteaceae	1–9
Leptomeria acida	Santalaceae	2
Leptospermum juniperinum	Myrtaceae	6
Leptospermum trinervium	Myrtaceae	1,2,3,5–9
Lepyrodia scariosa	Restionaceae	6
Leucopogon juniperinus	Ericaceae	1,2,4,5
Leucopogon microphyllus var. *microphyllus*	Ericaceae	1,2,4–7
Lindsaea linearis	Lindsaeaceae	1–6,8
Lomandra cylindrica	Lomandraceae	6
Lomandra glauca	Lomandraceae	1,3–9
Lomatia silaifolia	Proteaceae	1,4,5,7,8
Micrantheum ericoides	Picrodendraceae	4,6–9
Mirbelia rubioides	Fabaceae	6
Patersonia sericea	Iridaceae	1–9
Petrophile pulchella	Proteaceae	1–8
Philotheca reichenbachii	Rutaceae	1,5,6,9
Phyllanthus hirtellus	Phyllanthaceae	1–9
Phyllota phylicoides	Fabaceae	1–9
Pimelea linifolia subsp. *linifolia*	Thymeleaceae	5,6
Platysace linearifolia	Apiaceae	1–9
Poranthera ericoides	Euphorbiaceae	1,2,6–9
Pteridium esculentum	Dennstaedtiaceae	3
Pultenaea tuberculata	Fabaceae	1–9
Scaevola ramossisima	Goodeniaceae	1,2,3,5,8
Schizaea bifida	Schizaeaceae	1,5
Schoenus imberbis	Cyperaceae	6,8,9
Stackhousia viminea	Stackhousiaceae	3,6,8
Stylidium lineare	Stylidiaceae	6
Telopea speciosissima	Proteaceae	1,2,3
Tetratheca shiressii	Elaeocarpaceae	5,6,7
Thysanotus tuberosus subsp. *tuberosus*	Anthericaceae	1,3–6
Xanthorrhoea sp.	Xanthorrhoeaceae	1–9
Xylomelum pyriforme	Proteaceae	1,2,3
Xyris gracilis subsp. *gracilis*	Xyridaceae	6

SHORT COMMUNICATION

Assessing the conservation status of the grass *Elymus multiflorus* subsp. *kingianus* on Lord Howe Island, NSW

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Abstract

The grass *Elymus multiflorus* subsp. *kingianus* (family Poaceae) is considered to be endemic to Lord Howe and Norfolk Islands. We assessed the conservation status of the taxon on Lord Howe Island, New South Wales, and undertook targeted field surveys across three years and several seasons in suitable habitat, based on the single previously recorded location. We found the species occurrence on Lord Howe Island was highly restricted, with only a few plants found at 2 locations in close proximity. A number of exotic grasses pose a threat to the long-term conservation of the taxon, which was assessed as being eligible for listing as critically endangered under the *NSW Threatened Species Conservation Act 1995.*

***Cunninghamia* (2011) 12(2): 137–142**

Introduction

The assessment of the conservation status of plant and animal species is concerned with identifying those taxa that are considered to be most at risk of extinction. Recovery or threat management actions can then be undertaken to promote the long-term conservation of threatened species. The risk of extinction for plants and animals has been assessed using a variety of techniques ranging from specific decision rules (IUCN 1994, 2010, Keith 1998) to scoring systems based on attributes of species and threat (Millsap *et al.* 1990). Currently the most widely accepted scheme is the red list criteria developed by the World Conservation Union (IUCN 1994, IUCN 2010). This scheme underpins the criteria for listing species as threatened at the national level in Australia (*Environment Protection and Biodiversity Conservation Act* (*EPBC*) (see EPBC regulations 2000) and across several states (e.g. NSW *Threatened Species Conservation Act* 1995 (*TSC*) (see TSC Regulations 2010); Tasmanian *Threatened Species Protection Act 1995*).

The flora of Lord Howe Island (LHI) has a high level of endemism along with strong connections between mainland Australian, New Zealand and New Caledonian flora (Green 1994). There are also strong floristic affinities with Norfolk Island and several species co-occur on these two islands. *Elymus multiflorus* is a grass that occurs in coastal southeastern Australia, Norfolk Island, Lord Howe Island and New Zealand (De Lange *et al.* 2005). One form within the species (var. *kingianus*) has been recognised as a distinct variety (Green 1994, Conner 1994), and was recently accepted as being a subspecies (de Lange *et al.* 2005). *Elymus multiflorus* subsp. *kingianus* is considered to be endemic to Lord Howe Island in NSW and the Norfolk Island Group (Rabbit and Norfolk Islands) (Green 1994, de Lange *et al.* 2005). Conner (1994) distinguishes the Norfolk Island endemic *Elymus multiflorus* subsp. *kingianus* (now known to also occur on Lord Howe Island) from New Zealand plants by the multiplicity of small prickle teeth on the lemmas and on the glume. *Elymus multiflorus* subsp. *kingianus* has recently been listed as Critically Endangered on the *EPBC* Act following a review of the conservation assessment of plants from Norfolk Island (TSSC 2003). However, its status on Lord Howe Island (LHI) was somewhat uncertain. The taxon has been recorded on the Main island, under synonyms *Agyropogon scabrum* var. *scabrum* (Rodd & Pickard 1983) and *Agyropogon scabrum* (Pickard 1983), at only one location (Old Settlement Beach). It was considered rare and

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very uncommon by Pickard (1983). There have been no recorded sightings or collection records since that time. The ecology of the species on the island is also poorly known.

The IUCN Red List criteria (IUCN 2001) utilize three basic elements that may independently be used to assess the risk of extinction. These are: a) rates of decline; b) geographic range size; and c) population size and structure, including fluctuations and number of locations. In this paper, we undertook to assess the conservation status of *Elymus multiflorus* subsp. *kingianus* on LHI using IUCN red list criteria, along with NSW state *TSC* Act listing criteria and regulations. To do this we conducted survey work over several seasons across 3 years to verify the presence of *E. m. kingianus* on LHI, quantify its distribution and abundance and to identify any current threats to the taxon. We hoped that this would also help with the understanding of some basic life-history components of the ecology of the taxon.

Methods

Survey effort was concentrated on the one previously known location of Rodd & Pickard (1983), Old Settlement Beach, at the northern end of the island, along with all surrounding comparable habitat i.e., accessible basalt derived coastal margins in the northern half of the island (Fig. 1). Much of this habitat immediately adjacent to the settlement area is degraded by weeds (Hutton 2005). We conducted surveys over three years (2008, 2009, 2011) and three seasons

Table 1. Results of searches for *Elymus multiflorus* subsp. *kingianus* across likely habitat of Lord Howe Island (1971–2011). * = exotic plants. Collectors ANR= AN Rodd; TA= T Auld, AD= A Denham, IH= I Hutton, SB= S Bower, KM= K Macdonald, MS= M Stroud, CH= C Haselden, SN= S Nally, DM= D Milledge, HB= H Bower, IT= I Turnbull

Area	Coll.	Year	Season & date	Locations searched	*Elymus* abundance	Co-occurring taxa (* indicates exotics)	Threats
Dawsons Point coastline to Old Settlement Beach	ANR	1971	Autumn 21/03/1971	unknown	Occasional only	*Poa, Dianella, Melaleuca*	not recorded
	TA, AD, IH, SB	2008	Autumn 4/05/2008	Old Settlement Beach western end to point	3 plants	*Poa poiformis, Sporobolus virginianus, Melaleuca howeana, Cassinia tenuifolia, Dodonaea viscosa, *Paspalum* sp.	weeds
	TA, AD, IH, SB	2008	Autumn 4/05/2008	North Bay, eastern headland around to Dawsons Point	nil	*Sporobolus virginianus*, **S. africanus*, **Bromus cartharticus*, **Bromus diandrus, *Paspalum* spp.	weeds, erosion
	TA, AD, IH	2008	Spring 5/11/2008	Old Settlement Beach to North Bay around headland	1 plant	*Poa poiformis, Sporpbolus virginianus, Dichelachne crinita, *Sporobolus africanus, *Briza minor, *Sonchus oleraceus*	weeds
	SB, HB	2009	Autumn 4/5/2009	Old Settlement to Dawsons Point	27 plants (16 mature)	*Melaleuca howeana, Poa poiformis, Dodonaea viscosa, Cassinia tenuifolia, Rapanea platystigma, Dianella intermedia, *Bromus cartharticus, *Paspalum dilatatum, *Bromus* sp., **Ipomoea cairica*	weeds
	SB	2011	Spring 4/11/2011	Old Settlement to Dawsons Point	48 plants (9 mature)		weeds
	KM, MS, SB	2009	Summer 12–13/02/2009	Dawsons Point	2 (5 plants seen in 2010)	*Poa poiformis*, *Melanthera biflora, Leucopogon parviflorus, Melaleuca howeana, *Conyza bonariensis*	weeds, erosion
Dawsons Point ridge	TA, AD, IH, SB.	2008	Autumn 4/05/2008	Dawson ridge summit	nil	*Oplismenus hirtellus* under a 2–3m tree canopy.	none
Northern Hills	KM, MS, SB	2009	Summer 12–13/02/2009	Dawsons point to Kim's Lookout and Malabar track	nil		weeds
	SB, DM	2009	Winter 7–8/08/2009	North Head to Mt Eliza. Old Gulch to Curio Point	nil	*Commelina cyanea*, *Poa poiformis, Achyranthes aspera, Melaleuca howeana, *Bromus* sp., **Chloris gayana, *Asparagus asparagoides*	weeds
Roach Island	SB, CH, SN	2009	Summer 29/02/2009	Roach Island	nil	*Commelina cyanea*, *Achyranthes aspera, Poa poiformis, *Portulacca oleracea, *Bromus cartharticus, *Ipomoea cairica*	weeds
Rabbit Island	SB, HB, IT	2009	Autumn 15/04/2009	Rabbit Island	nil	*Commelina cyanea*, *Poa poiformis, Melaleuca howeana, *Chloris gayana, *Ipomoea cairica, *Bromus cartharticus*	weeds

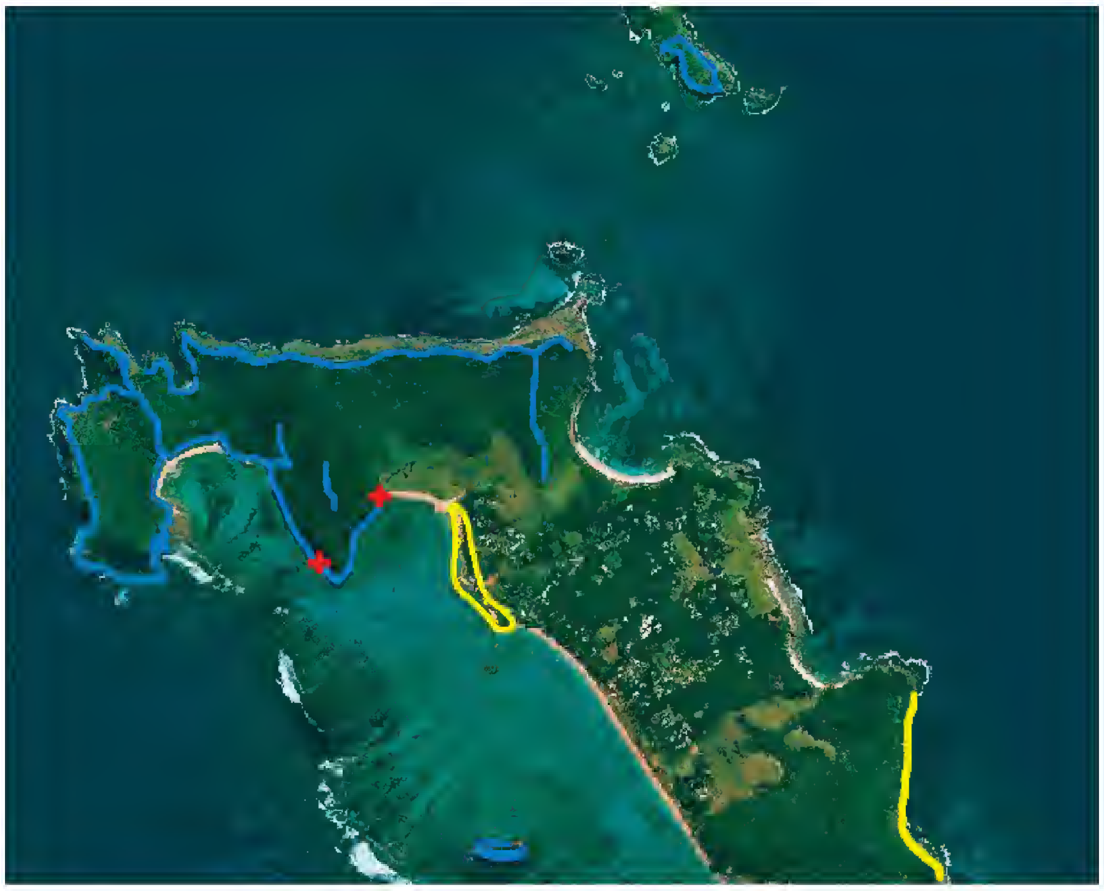

Fig. 1. Known locations of *Elymus multiflorus* subsp. *kingianus* (red x, near Dawson's Point and Old Settlement Beach) along with habitat searched (blue lines). Roach Island is at the top and Rabbit Island at the bottom of the Figure. The habitat on the east and southeastern side of Old Settlement Beach and from The Clear Place to Blinky Point is highly degraded by weeds (yellow line). The northern basalt cliffs are near vertical and inaccessible.

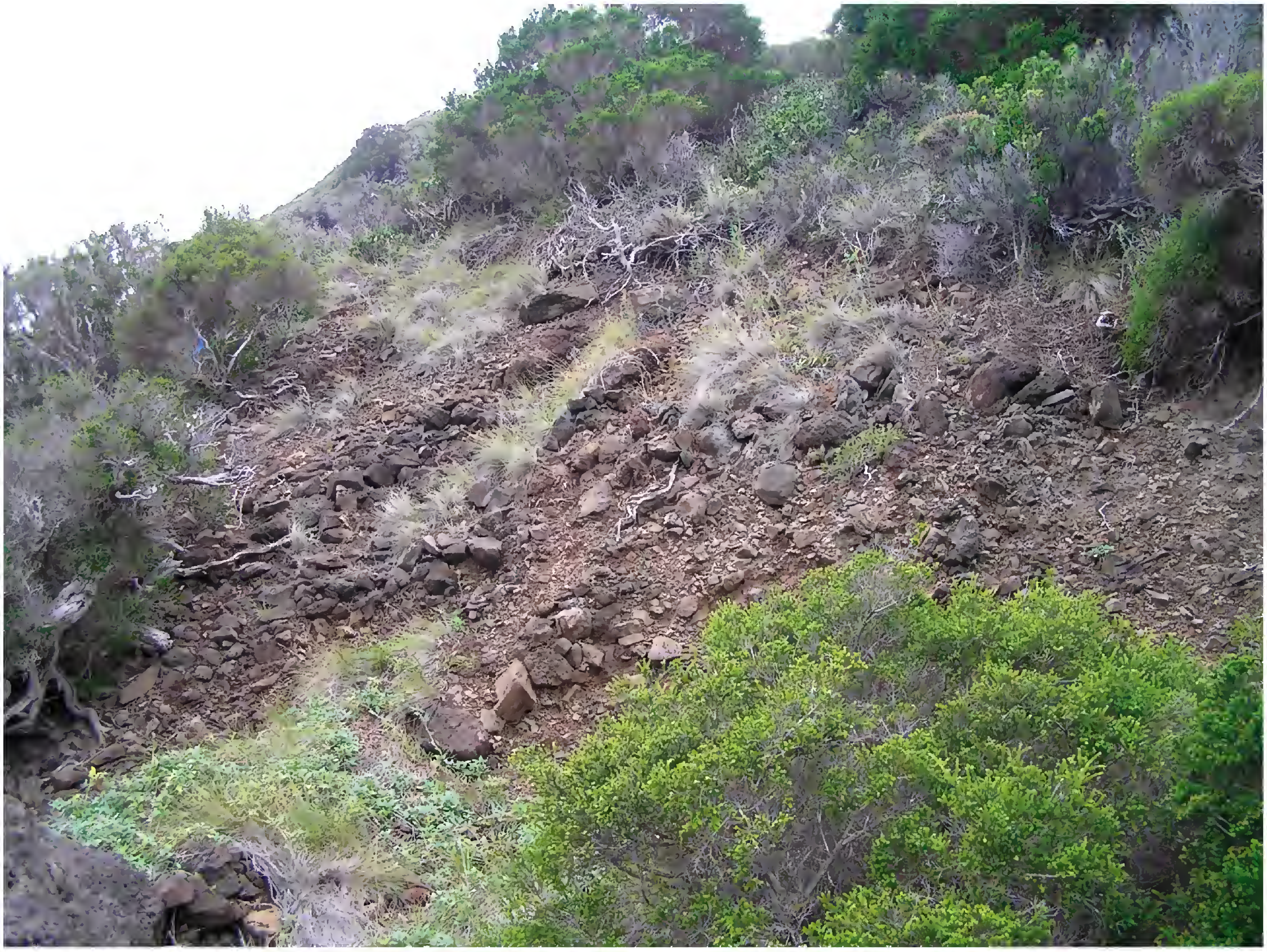

Fig. 2 Habitat for *Elymus multiflorus* subsp. *kingianus* at Dawsons point. Photo Sue Bower

Table 2. Assessment of conservation status of *Elymus multiflorus* subsp. *kingianus* on Lord Howe Island using IUCN red list criteria and NSW Threatened Species Conservation Act Regulation.

For each assessment five criteria or clauses determine if the taxon is threatened, and the degree of the threat, by placing it in a threat category (IUCN 2001, 2010, TSC 2010, NSW Scientific Committee 2010). The category used for the taxon once all five criteria/clauses have been considered, is the highest threat category reached by any one of the five criteria. Codes are DD– data deficient; CR –Critically Endangered; VU– Vulnerable.

Issue	IUCN Rule	IUCN	TSC Clause	TSC
Population Reduction[1]	A	DD	6	DD
Geographic Range[2]	B	CR via B1ab, B2ab	7	CR
Population Size[3]	C	CR via C2aii	8	CR
Small Population[4]	D	CR	9	CR
Extinction Probability[5]	E	DD	not applicable	
Very Restricted[6]	included in D		10	VU

1. There is insufficient information on the reduction of the size of the population. The one location identified by Rodd and Pickard (1983) remains extant.
2. The species is currently restricted to two small, but nearby, patches at the western end of Old Settlement Beach and at Dawsons Point. EOO and AOO are estimated to be 4 km^2, based on 2 x 2 km grid cells; the scale of assessment recommended by IUCN (2010). This is below the threshold for critically endangered. The habitat of the known locations and other potential habitat are occupied by a number of common natives *Poa, Sporobolus*, but also a number of weedy exotic grasses that may out compete *Elymus multiflorus* subsp. *kingianus*. Continuing decline is inferred as the habitat of the species is occupied by a number of weedy exotic grasses. For the species to be considered critically endangered under IUCN Criterion B, it must only occur in one location. The close proximity of the two known sites (a few hundred metres apart) would suggest that they are part of the one population and that the threats (weeds) will affect both sites.
3. The population is estimated to be less than 50 mature individuals. Continuing decline is inferred as the habitat of the species is occupied by a number of weedy exotic grasses that may out compete *Elymus multiflorus* subsp. *kingianus*. As no subpopulation is estimated to contain more than 50 mature individuals, the species would be assessed as Critically Endangered under IUCN.
4. The population size is extremely small and estimated to be less than 50 mature individuals (which is equivalent to extremely low under TSC Regulation 2010, NSW Scientific Committee 2010).
5. There is insufficient data for quantitative analysis. Criterion E is data deficient.
6. The restricted nature of the distribution combined with threats and stochastic events would allow the taxon to be assessed as vulnerable under this clause.

(autumn and summer 2008, summer, autumn and winter 2009, spring 2011). All observers were initially shown plants at a known location to standardise identification. We searched along the base of littoral rainforest just above bare rocks on the coastal shore, where exposure has led to a slightly more open habitat, as it is in this zone that there is available habitat for the species. This included erosion scarps providing areas of potential habitat for recruitment. We also searched the nearby summit of Dawson ridge where we examined areas of exposed rocky habitats with grass or sedge cover, along with other exposed areas on walking tracks nearby. Where plants were located we counted the number of individuals, recorded vegetation composition, including weedy species and recorded any threats to the species. Additional sites along the sea cliffs on the eastern side of the island above the Clear Place to Blinky Point were also searched.

Results

We successfully re-located *Elymus multiflorus* subsp. *kingianus* at the western end of Old Settlement Beach, the previously known location of Rodd & Pickard (1983), and at one nearby additional location at Dawsons Point (Fig. 1, Table 1). Both locations were in comparable habitat i.e. the zone between exposed basalt-derived cliffs and boulders near the water's edge, and the closed canopy of littoral rainforest upslope. This zone was dominated by shrubs, grasses and herbs, and was a mixture of closed shrub and open areas, either with some vegetation cover or exposed soil with little vegetation cover. At one site, *Elymus* plants occurred on the edges of an erosion scarp (10m ASL) and at the base of the erosion slip (>2m ASL). The width of the suitable habitat varied from <1 to a few metres, but was more or less continuous around the shoreline on the lagoon side of the northern part of the Main island. Most of the habitat of coastal rocky shoreline that was searched was not occupied by *Elymus* and no plants were found in any exposed rocky patches on Dawsons Ridge. The habitat on the east and southeastern side of Old Settlement Beach is highly degraded by weeds and past clearing of the littoral rainforest (Fig. 1). No plants were found in the habitat along the extent of the sea cliffs above the Clear Place to Blinky Point; this habitat is severely degraded by *Asparagus aethiopicus* (Ground asparagus). Remnant native *Poa poiformis* grass communities are also compromised by severe weed competition in this area.

In all searches, from 1 up to 48 plants of *Elymus multiflorus* subsp. *kingianus* were seen, abundance depending on seasonal and annual variations (Table 1); it is inferred there are fewer than 50 mature plants in total. The main threat to *Elymus multiflorus* subsp. *kingianus* plants are exotic weeds that can occupy the habitat of the species and are likely to compete for space. There is also localised erosion that may result in the loss of individuals, but may in the long term also allow future recruitment into open patches. Excessive dry periods may threaten plants that occur in small crevices on cliffs. Storm surge may threaten low-lying plants. Other possible threats are previous damage from goats (now eradicated) and seed and seedling predation by exotic rodents.

The conservation status of *Elymus multiflorus* var. *kingianus* on LHI was assessed as critically endangered under both IUCN and the NSW threatened Species Conservation Act Regulations (Table 2).

Both estimates of range size (extent of occurrence EOO, and area of occupancy AOO) are estimated to be 4 km^2, based on 2 x 2 km grid cells; the scale of assessment recommended by IUCN (2010).

Discussion

Elymus multiflorus subsp. *kingianus* was assessed as critically endangered in NSW under both IUCN red list criteria and *TSC* Act regulations (2010). Its very restricted habitat, low numbers and ongoing threats from weeds were the main determinants of this threats status. At a global scale, incorporating the locations in the Norfolk Island group, low population numbers, ongoing weed threats (NI and LHI) and grazing (NI, Director of National Parks 2010) would also likely lead to an assessment of critically endangered.

We found that *Elymus multiflorus* subsp. *kingianus* had persisted on Lord Howe Island in one known location for over 25 years, even though the number of plants we detected was very low (1–48, depending on season and year). We could only locate one other nearby location (Fig. 1) with 2 plants in 2009 (up to 5 in 2010). The habitat for the species extends beyond these two sites and *Elymus* may potentially occupy some of this habitat in other years. There was some variation in abundance across years and seasons at the original known site which likely reflects seasonal dieback and variation in seasonal conditions controlling recruitment and growth. Further searching in favourable seasons is recommended. Much of the habitat is in good condition, but there are a number of weedy species that pose a threat to *Elymus* and other co-occurring native species (Table 1). These are exotic grasses, herbs and creepers that are disturbance opportunists, particularly *Sporobolus africanus*, *Bromus cartharticus, Bromus diandrus* and *Paspalum* spp. that have the potential to successfully occupy the *Elymus* habitat (see Fig. 2) and exclude native taxa.

Our data suggest that individual plants are short-lived and there may be a rapid turnover in plant numbers at any one location (Table 1). This short-lived pattern corresponds with other reports on the ecology of *Elymus* spp. suggesting it is a short-lived perennial (Benson & McDougall 2005). McIntyre *et al*. (1995) suggest that there is no vegetative spread in the closely related *Elymus scaber* and this also appears likely in *Elymus multiflorus* subsp. *kingianus*. McIntyre *et al*. (1995) also suggest that the dispersal unit in *Elymus scaber* is adhesive, most likely through large awns. Adhesive characteristics are likely to promote dispersal on mammals. Awns in *Elymus multiflorus* subsp. *kingianus* on Norfolk Island are up to 17mm long (Conner 1994) and combined with the small prickle teeth on the lemmas and on the glumes may aid adhesion. As mammals are absent on Lord Howe Island (except for a bat) the ability of *Elymus multiflorus* subsp. *kingianus* to move to different patches of potential habitat around its current location is unknown. The species would need to maintain a persistent soil seed bank to take advantage of gaps in its habitat and to recruit each year. Currently factors controlling the timing and magnitude of germination are unknown, but seed dormancy has been observed in *Elymus scaber* (Maze *et al*. 1999), along with water and temperature impacts on germination.

A permanent monitoring plot and rodent bait station was established by the Lord Howe Island Board, and some individual plants were tagged in 2011.

Ongoing management of *Elymus multiflorus* subsp. *kingianus* should focus on monitoring the existing locations and where possible, other nearby available habitat to detect season and magnitude of germination in any one year. Some weed control may be necessary where any aggressive invasion of habitat occurs. Maintenance of bait stations should also be ongoing.

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Floristic variability, distribution and an extension of range for the endangered Pittwater Spotted Gum Forest, Central Coast, New South Wales

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Abstract: Multivariate analysis and non-metric multidimensional scaling have been used to compare the floristic composition of vegetation dominated by Spotted Gum, *Corymbia maculata,* and Grey Ironbark, *Eucalyptus paniculata,* from the Pittwater and Gosford local government areas, and elsewhere on the Central Coast of New South Wales. When placed within a regional context, vegetation from both local government areas is strongly related, and is sufficiently distinct (at 30% similarity) from other areas of Spotted Gum-Ironbark vegetation within the region to warrant their collective consideration as Pittwater Spotted Gum Forest (PSGF), an Endangered Ecological Community in New South Wales. Within Pittwater local government area, two forms of PSGF are present: a dryer, more widespread form present on ridges and slopes, and a moister form with strong littoral rainforest affinities confined to gullies. Gosford local government area supports only the dryer form. Extant and pre-1750 mapping of PSGF from both local government areas (excluding Ku-ring-gai Chase National Park) shows a total extant distribution of 227 ha, and since 1750 an estimated loss of 727 ha (76%) from the original 954 ha. PSGF is present in two secure conservation reserves: Ku-ring-gai Chase National Park (~40 ha, Thomas & Benson 1985) and Bouddi National Park (~30 ha, Bell 2009), totaling ~ 70 ha. Despite this, much of the extant distribution lies at the reserve-urban interface, and is subject to ongoing threats of regular low-intensity fire, unmitigated clearing and weed invasion.

Key words: threatened community, Pittwater Spotted Gum Forest, classification, mapping

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Introduction

Within New South Wales, forests and woodlands dominated by Spotted Gum (*Corymbia maculata, Corymbia variegata, Corymbia henryi*) and various ironbark eucalypts (eg: *Eucalyptus paniculata, Eucalyptus siderophloia, Eucalyptus fibrosa, Eucalyptus crebra*) are widespread. Indeed, in his overview of the native vegetation of New South Wales, Keith (2004) outlined four classes of vegetation where such a combination of canopy species is characteristic (Southern Lowland Wet Sclerophyll Forests, Coastal Valley Grassy Woodlands, Clarence Dry Sclerophyll Forests, Hunter-Macleay Dry Sclerophyll Forests), encompassing almost the entire coastal strip. Spotted Gum is also a commercially important tree species, having a long history of timber production (FCNSW 1989; Boland et al 2006; Wrigley & Fagg 2010) and in the production of honey (Clemson 1985; Law & Chidel 2008).

Along much of the coastline of New South Wales, *Eucalyptus paniculata* (Grey Ironbark) is the most widespread ironbark species, ranging from Coffs Harbour to Bega, and rarely extending more than 80km inland (Boland et al. 2006). Together with Spotted Gum, both species are characteristic of coastal forests in the South, Central and North Coast botanical subdivisions, as evidenced by their prominence in vegetation classifications for these areas (eg: NSW NPWS 2000; Gellie 2005; Somerville 2009). Pidgeon (1937), in her seminal work on the vegetation of the New South Wales Central Coast (Hunter Valley to the Lower Shoalhaven Valley), described an '*Eucalyptus maculata–Eucalyptus paniculata* Association' extending sporadically from Lake Macquarie south to Batemans Bay. However, there has been little investigation to date regarding how geographically isolated stands of vegetation co-dominated by *Corymbia maculata* and *Eucalyptus paniculata* may differ, or indeed whether or not the same association of co-occurring species is present.

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Pittwater Spotted Gum Forest (PSGF) was first named and described for parts of northern Sydney by Benson and Howell (1990a, 1994). It was subsequently listed as an Endangered Ecological Community in New South Wales under the *Threatened Species Conservation Act 1995*, where it was described as occurring on shale-derived soils in high rainfall areas, on lower hillslopes of the Newport formation along Barrenjoey Peninsula and on the western shores of Pittwater (NSW Scientific Committee 1999). Benson and Howell (1994) delineated two forms of the forest, but both were dominated by a canopy of *Corymbia maculata*. On dryer sites, *Allocasuarina littoralis, Dodonaea triquetra, Platylobium formosum, Macrozamia communis* and *Pultenaea flexilis* comprised the sparse small tree and shrub layer, over an open ground cover of herbs and graminoids. In the more sheltered aspects, the small tree layer included *Elaeocarpus reticulata* and *Pittosporum undulatum*, with a mesic understorey of *Diospyros australis, Synoum glandulosum, Cassine australis* var. *australis, Melicope micrococca, Commersonia fraseri, Alphitonia excelsa* and *Livistona australis*. Ferns (*Doodia aspera, Adiantum aethiopicum*) and vines (*Cissus hypoglauca, Pandorea pandorana, Cayratia clematidea, Geitonoplesium cymosum, Eustrephus latifolius*) were also common.

Holden (2001) later undertook an investigation into aspects of the PSGF as part of an Honours degree at the University of New South Wales. He found that at that time only 51 ha of remnant PSGF remained. Increases in soil nutrients as a result of urban expansion was suggested by Holden (2001) as the main cause behind the higher abundance of the mesomorphic species *Pittosporum undulatum, Glochidion ferdinandi* and *Elaeocarpus reticulatus* within the former open forests. Other threats, such as infrequent and low intensity fire, and increasing urban development were also discussed in the context of future management.

Recent vegetation survey and classification projects undertaken on behalf of local government have gathered a considerable amount of additional data on known and potential PSGF (hereafter termed candidate-PSGF). A numerical classification of the native vegetation of the Gosford local government area identified patches of vegetation around the northern foreshores of Broken Bay which supported many of the features of PSGF (Bell 2009). A similar project was undertaken within the Pittwater local government area (Bangalay & Eastcoast Flora Survey 2011), and it is appropriate that a review of the community and its conservation status now be made.

This paper examines the relationship between areas supporting forests dominated by *Corymbia maculata* and *Eucalyptus paniculata,* from both the Pittwater and Gosford local government areas. It forms part of a broader study being undertaken on Spotted Gum-Ironbark forests within the Central Coast and Hunter Valley region of New South Wales. In part, this study attempts to answer recent questions over the significance of vegetation considered as candidate-PSGF within the Gosford LGA, but also raises new questions on the relationships of other more geographically distinct stands of vegetation elsewhere in the State.

Study Area

The Pittwater (excluding Ku-ring-gai Chase National Park) and Gosford local government areas (LGAs) north of Sydney, New South Wales, form the core area for this study (Figure 1). Floristic regional data have also been drawn from the Hunter Valley and Central Coast (Hunter-Central Rivers Catchment Management Area: CMA) for comparative analysis. Both Pittwater and Gosford LGAs fall within a warm temperate climatic zone, with a maritime influence near the coast, and experience warm wet summers and cool dry winters. The majority of Pittwater LGA and the eastern parts of Gosford LGA occur within the 1,100 to 1,300 isohyet range. Rainfall occurs throughout the year, although is generally higher in summer and autumn, largely due to onshore winds, and with a mean annual rainfall of 1212 mm at Observatory Hill (Bureau of Meteorology 2010).

Geologically, Pittwater and Gosford LGAs occur on Triassic sediments of the Sydney Basin, an extensive geological feature comprising horizontal beds of sandstones and shales, and covering 36 000 km^2 of land from Batemans Bay in the south to the Hunter River in the north (Bembrick et al 1980). The two LGAs are separated only by an expanse of ocean known as Broken Bay. Soil landscapes in both areas have been studied in some detail (Chapman & Murphy 1989; Murphy 1993), and span coluvial, residual, erosional, fluvial, aeolian, marine, estuarine, swamp and disturbed landscapes. In general, soils derived from the younger Triassic Hawkesbury Sandstone overlie those derived from the Triassic Narrabeen Series.

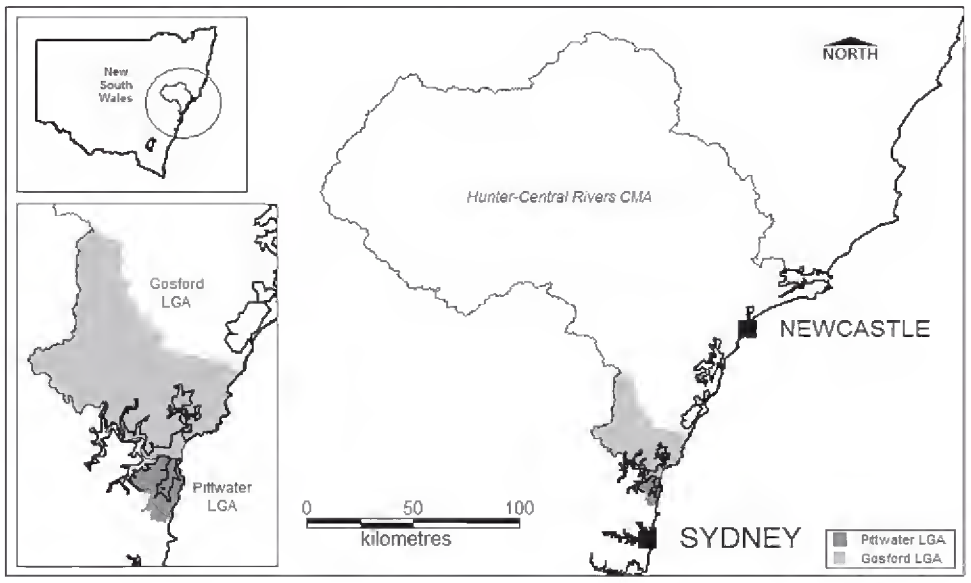

Fig. 1. Pittwater and Gosford local government areas, and the Hunter-Central Rivers CMA region.

A number of vegetation classifications have been developed for both Pittwater and Gosford LGAs over recent decades. Among these are the sub-regional studies of Benson and Howell (1994), NSWNPWS (2000), McCauley et al. (2006), Somerville (2009) and NSWDECCW (2010), and the recent LGA-focused studies of Bell (2009) and Bangalay & Eastcoast Flora Survey (2011). All provide good overviews of the diversity of vegetation present within the study area.

Methods

Sampling

Existing data collected by the authors and adhering to the New South Wales standard (Sivertsen 2010) were extracted to form a base analysis dataset. New data was collected to augment existing data and to target poorly sampled locations. Areas supporting stands of *Corymbia maculata* as a dominant or co-dominant species in the canopy (with or without *Eucalyptus paniculata*) were preferentially selected for detailed survey in both the Gosford and Pittwater LGAs. Reference was made to the mapping of Holden (2001) for Pittwater LGA and Bell (2009) for Gosford LGA as a guide to site selection. Given constraints imposed by private land ownership, the majority of locations sampled were Council-owned reserves. No new sampling was completed within Ku-ring-gai Chase National Park. All sample plots were of 0.04ha (20 x 20m) and located within locally homogeneous stands of vegetation. Modified (1–6 scale) Braun-Blanquet cover abundance scores (Braun Blanquet 1928) were applied to all vascular plant species recorded within each plot. Plant nomenclature followed Harden (1990–1993) and revisions subsequently accepted by the National Herbarium of New South Wales.

Classification and analysis

Numerical classification (UPGMA cluster analysis & nMDS ordination) on collected data was performed using *Primer* V6 (Clarke & Gorley 2006), utilizing the group averaging strategy, the Bray-Curtis association measure and a Beta value of –0.1. Data were analysed initially within a larger regional dataset (Hunter-Central Rivers CMA) of sample plots dominated by *Corymbia maculata* and various ironbark species, to identify those samples where candidate-PSGF may potentially occur. Newly collected data was analysed to provide a regional context to the PSGF dataset. The SIMPER routine in Primer was used to generate diagnostic species lists for each defined floristic group.

Mapping

Mapping the distribution of candidate-PSGF combined interpretation of digital orthorectified imagery and on-screen digitising of vegetation boundaries with extensive ground truthing. In both LGAs, exhaustive vehicle and foot traverses were undertaken (excluding some private lands), and data on dominant plant species within each major strata (canopy, shrub, ground) recorded at regular locations with a hand-held GPS unit. Collected data was imported into a GIS system and used to inform and classify map polygons, using dominant species as indicators for vegetation communities. Nomenclature for communities followed Bell (2009) for Gosford LGA, and Bangalay & Eastcoast Flora Survey (2011) for Pittwater LGA. Candidate-PSGF included Wagstaff Spotted Gum Forest (Unit 15b) in Gosford, and Coastal Moist Spotted Gum Forest (Unit S_WSF11) and Coastal Dry Spotted Gum Forest (Unit S_DSF25) in Pittwater. To estimate the pre-1750 distribution of candidate-PSGF, the same data collection techniques were used across cleared and urban landscapes to record remnant canopy species, which were later allocated to vegetation communities. Within the GIS, this data was used in combination with elevation and soils data layers to interpolate likely pre-disturbance boundaries. Further details on this process can be found in Bell (2009) and Bangalay & Eastcoast Flora Survey (2011).

Results

Classification

A total of 23 plots were sampled in Gosford and Pittwater LGAs, within known and candidate-PSGF. Within the regional dataset of nearly 500 samples, these plots grouped together to form two coherent clusters, at ~30% similarity (Figure 2). These 23 samples of candidate-PSGF were extracted from the main database to form a subset of sites for further investigation.

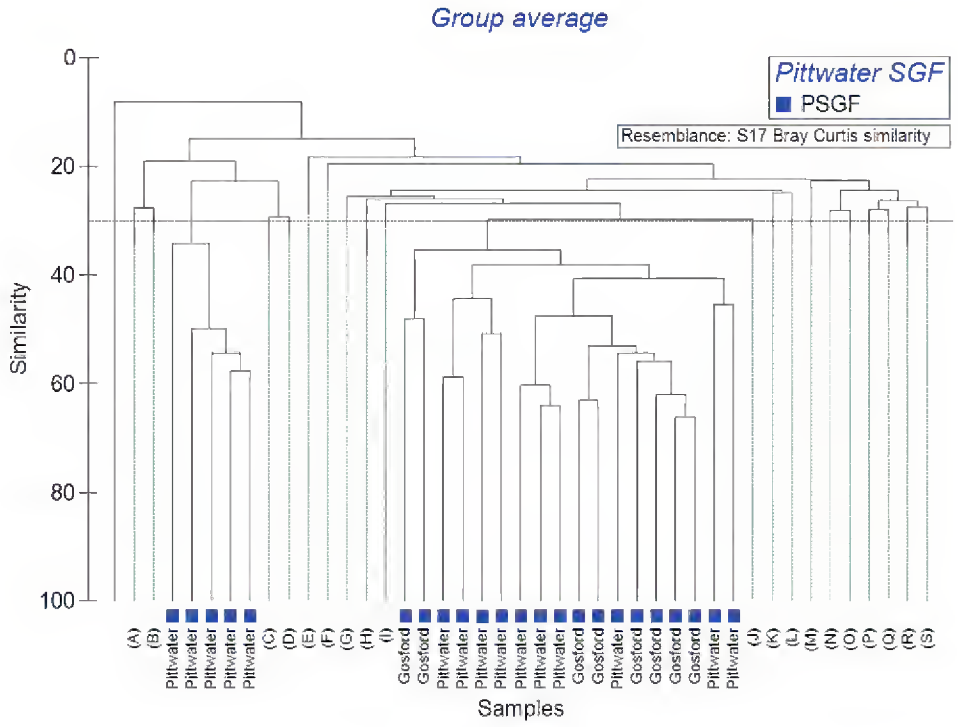

Fig. 2. Site dendrogram (partially collapsed at 30% similarity) showing the position of candidate-PSGF plots (2 groups, with LGA labels) within a regional dataset of 489 Spotted Gum samples (Bray-Curtis association measure). Floristic groups defined at 30% similarity (horizontal dotted line).

Table 1 Species comprising the upper 90% of diversity within Group A and Group B, arranged in decreasing order of importance.

Group A – Average similarity: 43.58

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Corymbia maculata	3.67	4.61	4.82	10.59	10.59
Entolasia stricta	1.94	2.29	2.98	5.26	15.85
Pteridium esculentum	2.11	1.90	1.15	4.35	20.20
Dianella caerulea	1.61	1.80	2.99	4.13	24.33
Pandorea pandorana	1.78	1.75	1.92	4.01	28.34
Lomandra longifolia	1.50	1.58	2.99	3.63	31.97
Themeda australis	1.61	1.52	1.78	3.48	35.44
Eustrephus latifolius	1.44	1.48	2.09	3.40	38.84
Eucalyptus paniculata subsp. *paniculata*	1.67	1.40	1.00	3.21	42.05
Pratia purpurascens	1.39	1.32	1.51	3.04	45.09
Macrozamia communis	1.61	1.23	0.84	2.83	47.92
Imperata cylindrica var. *major*	1.56	1.21	1.07	2.77	50.69
Notelaea longifolia forma *longifolia*	1.33	0.96	1.06	2.21	52.90
Entolasia marginata	1.17	0.84	0.93	1.94	54.84
Elaeocarpus reticulatus	1.22	0.84	0.91	1.94	56.78
Pultenaea flexilis	1.44	0.83	0.50	1.91	58.68
Geitonoplesium cymosum	1.11	0.80	0.93	1.84	60.52
Persoonia linearis	0.83	0.80	1.18	1.83	62.35
Smilax glyciphylla	1.22	0.80	0.77	1.83	64.18
Lomandra multiflora subsp. *multiflora*	1.06	0.78	0.89	1.79	65.97
Pseuderanthemum variabile	1.00	0.75	0.93	1.73	67.71
Glochidion ferdinandi var. *ferdinandi*	1.00	0.75	0.94	1.71	69.42
Pittosporum undulatum	1.06	0.72	0.78	1.66	71.08
Panicum simile	0.89	0.71	0.99	1.62	72.71
Pomax umbellata	0.83	0.69	1.00	1.57	74.28
Billardieria scandens	0.78	0.61	0.84	1.39	75.67
Acacia ulicifolia	0.94	0.60	0.67	1.39	77.06
Caesia parviflora	0.72	0.57	0.86	1.32	78.37
Allocasuarina littoralis	1.06	0.56	0.46	1.29	79.66
Hardenbergia violacea	0.89	0.56	0.72	1.28	80.94
Glycine clandestina	0.89	0.52	0.70	1.18	82.12
Allocasuarina torulosa	1.06	0.45	0.42	1.04	83.16
Digitaria parviflora	0.61	0.30	0.54	0.68	83.84
Breynia oblongifolia	0.50	0.29	0.55	0.67	84.51
Desmodium rhytidophyllum	0.56	0.29	0.55	0.67	85.17
Phyllanthus hirtellus	0.56	0.27	0.46	0.62	85.80
Lomandra filiformis subsp. *filiformis*	0.56	0.26	0.47	0.59	86.38
Persoonia levis	0.50	0.26	0.47	0.59	86.97
Lomandra confertifolia subsp. *pallida*	0.67	0.25	0.31	0.57	87.54
Microlaena stipoides var. *stipoides*	0.61	0.25	0.45	0.57	88.11
Myrsine variabilis	0.56	0.24	0.46	0.56	88.66
Eucalyptus umbra	0.78	0.22	0.31	0.51	89.17
Acacia implexa	0.56	0.22	0.37	0.50	89.67
Cassytha pubescens	0.56	0.19	0.38	0.45	90.12

Group B – Average similarity: 45.18

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Pittosporum undulatum	2.80	3.55	4.37	7.85	7.85
Geitonoplesium cymosum	2.00	3.11	5.56	6.89	14.74
Morinda jasminoides	2.40	3.11	5.56	6.89	21.63
Corymbia maculata	2.80	2.93	1.05	6.48	28.11
Livistona australis	2.60	2.84	1.23	6.29	34.40
Sarcopetalum harveyanum	1.80	2.00	2.68	4.44	38.84
Eustrephus latifolius	1.60	1.91	1.12	4.23	43.07
Smilax glycinoides	1.40	1.71	3.19	3.78	46.85
Wilkea huegeliana	1.60	1.45	1.07	3.20	50.05
Cissus hypoglauca	1.80	1.34	0.62	2.98	53.03
Notelaea longifolia forma *longifolia*	1.40	1.33	0.98	2.95	55.98
Calochlaena dubia	2.40	1.30	0.52	2.87	58.85
Oplismenus imbecillus	1.60	1.23	1.09	2.71	61.56
Ficus coronata	1.40	1.21	0.90	2.69	64.25
Synoum glandulosum subsp. *glandulosum*	1.20	1.15	1.05	2.55	66.80
Glochidion ferdinandi var. *ferdinandi*	1.60	1.09	0.95	2.41	69.20
Gymnostachys anceps	1.00	1.00	1.13	2.21	71.41
Pittosporum revolutum	0.80	1.00	1.13	2.21	73.63
Pandorea pandorana	1.20	1.00	1.07	2.21	75.84
Commelina cyanea	1.40	1.00	1.00	2.21	78.05
Macrozamia communis	0.80	0.96	1.12	2.11	80.16
Cayratia clematidea	1.40	0.76	0.62	1.68	81.84
Microlaena stipoides var. *stipoides*	1.40	0.76	0.62	1.68	83.52
Lomandra longifolia	1.00	0.62	0.56	1.38	84.90
Stephania japonica var. *discolor*	1.00	0.59	0.55	1.31	86.21
Eucalyptus paniculata subsp. *paniculata*	1.20	0.58	0.60	1.28	87.49
Adianthum aethiopicum	0.80	0.56	0.61	1.24	88.73
Brachychiton acerifolius	0.60	0.53	0.61	1.17	89.90
Passiflora herbertiana var. *herbertiana*	0.80	0.53	0.61	1.17	91.06

Both the cluster analysis (Figure 3) and the nMDS ordination (Figure 4) performed on this subset of data resulted in two definable floristic groups at 30% similarity. The eighteen samples comprising Group A were a combination from both Gosford (8) and Pittwater (10) LGAs. Group B comprised samples only from Pittwater LGA.

The SIMPER routine identified species that characterise each of the two Groups, and those species which contribute the upper 90% of floristic diversity to each Group are shown in Table 1. Between the two groups, there was only one species (*Corymbia maculata*) shared within the upper 50% of diversity, and a further 11 shared within the upper 90% of diversity (the tree *Eucalyptus paniculata* subsp. *paniculata,* the shrubs *Glochidion ferdinandi* var. *ferdinandi, Notelaea longifolia* forma *longifolia* and *Pittosporum undulatum,* the vines *Eustrephus latifolius, Geitonoplesium cymosum, Pandorea pandorana,* and *Smilax glycinoides,* the cycad *Macrozamia communis,* the graminoid *Lomandra longifolia,* and the grass *Microlaena stipoides* var. *stipoides*). Group A is dominated by dryer understorey species and grasses, and may be designated as a 'dry' form of PSGF, while Group B is dominated by mesic species of small trees and shrubs, and can be termed 'moist PSGF'. Species found to be unique to either of the two defined Groups are shown in Table 2, with many more unique species being found in the dry PSGF.

Mapping

Figure 5 shows the extant and estimated pre-1750 distribution of candidate-PSGF within the Gosford and Pittwater LGAs (excluding Ku-ring-gai Chase National Park). Collectively, the two areas currently support 227 ha of this vegetation community (157 ha dry PSGF and 69 ha moist PSGF), while percentage loss figures combined, based on pre-1750 mapping, total 728 ha from an original 954 ha (Table 3). Candidate-PSGF in the Pittwater LGA has clearly been subjected to a higher level of clearing for development than it has in Gosford. Within Ku-ring-gai Chase National Park, previous estimates have shown there to be ~40 ha (Thomas & Benson 1985), but this has not been substantiated during the current study and has been excluded from the above figures.

Discussion

Elsewhere in New South Wales, vegetation associations dominated by *Corymbia maculata* and *Eucalyptus paniculata* occur sporadically along some parts of the coastal zone. Pidgeon (1937) was perhaps the first to recognize a *Corymbia maculata-Eucalyptus paniculata* association, describing it for areas of Lake Macquarie, Pittwater, the Illawarra, Nowra and in the Batemans Bay district. None of these areas have yet been examined and compared in detail, although all have been included in regional or sub-regional classification studies. In more recent times, vegetation dominated by *Corymbia maculata* and *Eucalyptus paniculata* in the Lake Macquarie area has been included in broader units, such as the Coastal Foothills Spotted Gum-Ironbark Forest of NSWNPWS (2000) or the Spotted Gum/ Grey Ironbark Dry Open Forest of the Central Coast and Lower Hunter of Somerville (2009). In the Illawarra, NSWNPWS (2002) describe Spotted Gum Open Forest from a single location, and state that it has all but disappeared from the Illawarra plain and that profiling its composition is difficult. Gellie (2005) describes both a dry (Coastal Lowlands Cycad Dry Shrub Dry Forest) and moist (Northern Foothills Moist Shrub Forest) for the South Coast, and both support strong similarities to candidate-PSGF. Based on descriptions available for all of these areas, the *Corymbia maculata–Eucalyptus paniculata* association of Pidgeon (1937) is clearly a tangible vegetation type, with

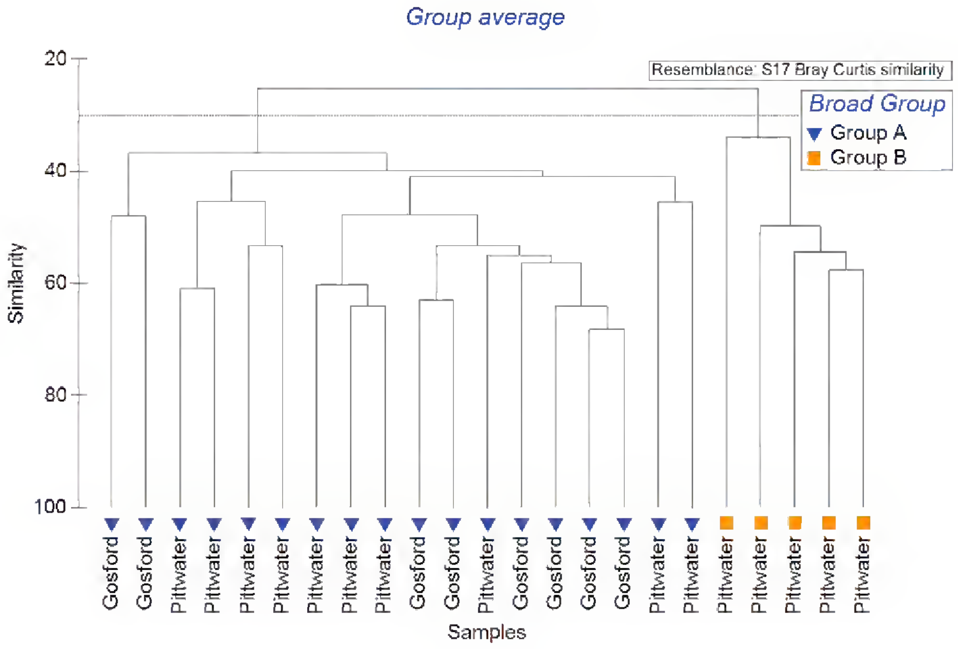

Fig. 3. Site dendrogram showing the relationship between all sample plots comprising candidate-PSGF (Bray-Curtis association measure). Floristic groups defined at 30% similarity (horizontal dotted line).

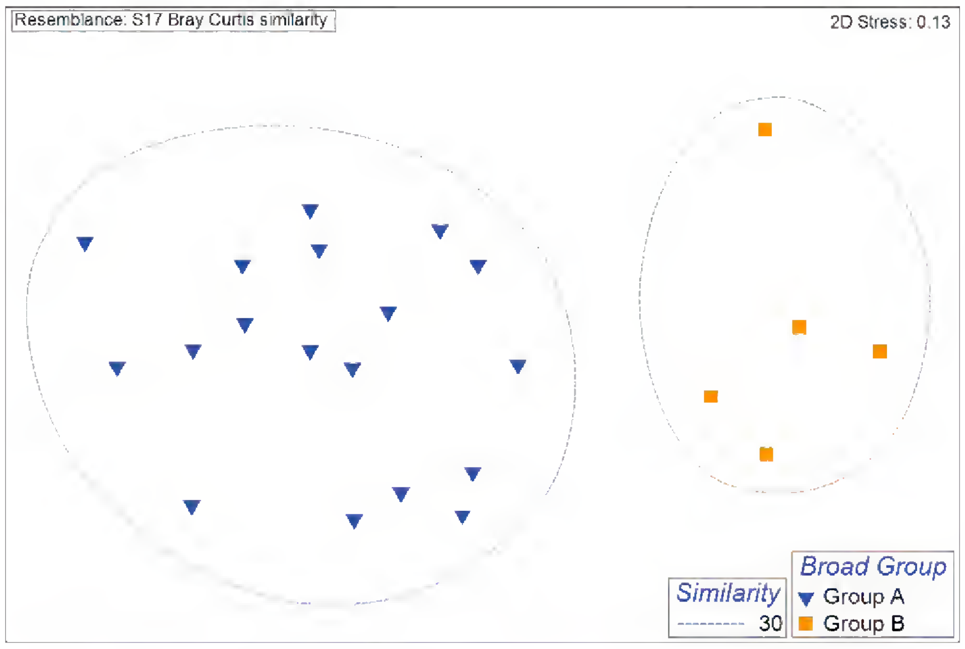

Fig. 4. nMDS plot showing the relationship between sample plots comprising candidate-PSGF, overlain with cluster analysis groups from Figure 3 (Bray-Curtis association measure). Stress = 0.13.

Table 2 Species unique to Group A or Group B, arranged by growth habit.

Habit	Group A (dry)	Group B (moist)
Tree	*Angophora costata* *Corymbia gummifera*	*Acmena smithii* *Eucalyptus scias* subsp. *scias* *Ficus coronata*
Shrub	*Acacia implexa* *Acacia ulicifolia* *Acrotricha divaricata* *Allocasuarina littoralis* *Hakea sericea* *Hibbertia aspera* subsp. *aspera* *Persoonia levis* *Persoonia linearis* *Podolobium ilicifolium* *Polyscias sambuccifolia* *Pultenaea flexilis*	*Myrsine howitiana* *Synoum glandulosum* subsp. *glandulosum* *Trema tomentosa* var. *viridis*
Herb	*Brachyscome angustifolium* var. *angustifolium* *Caesia parviflora* *Desmodium gunnii* *Desmodium rhytidophyllum* *Hibbertia diffusa* *Hybanthus monopetalus* *Phyllanthus hirtellus* *Pomax umbellata* *Pratia purpurascens* *Schellhamera undulata* *Wikstroemia indica*	-
Ferns	-	*Doodia aspera*
Vines	*Billardiera scandens* Cassytha pubescens *Hardenbergia violacea* *Hibbertia dentata*	*Rubus parvifolius*
Grass	*Aristida vagans* *Cymbopogon refractus* *Digitaria parviflora* *Echinopogon caespitosus* *Entolasia stricta* *Imperata cylindrical* var. *major* *Oplismenus aemulus* *Panicum simile* *Poa affinis* *Themeda australis*	-
Sedge	-	*Carex longebrachiata* *Gahnia melanocarpa*
Graminoid	*Lomandra confertifolia* subsp. *pallida* *Lomandra filiformis* subsp. *filiformis* *Lomandra glauca* *Lomandra multiflora* subsp. *multiflora* *Xanthorrhoea arborea* *Xanthorrhoea macronema*	-

perhaps the PSGF comprising but one part of a wider ranging complex.

At the time of its listing as endangered on the *Threatened Species Conservation Act 1995* in the late 1990s, the Pittwater Spotted Gum Forest was thought to be restricted to Pittwater LGA (NSW Scientific Committee 1999). With increasing survey and mapping effort being expended in the coastal zone of New South Wales, it is not surprising that potentially similar stands of vegetation in other areas would be encountered. The current study has shown that vegetation described for parts of the neighbouring Gosford LGA is indeed floristically more similar to that found in Pittwater than it is to other areas of Spotted Gum-Ironbark vegetation within the Hunter-Central Rivers CMA region (nor elsewhere in the adjacent Sydney Metropolitan CMA: NSWDECCW 2010). Although not extensive, the stands of candidate-PSGF reported on here from the Gosford LGA add considerably to the known distribution of this community, and perhaps more importantly, add to the extent in formal conservation tenure. Pittwater Spotted Gum Forest (here defined) is currently represented within two secure conservation reserves: ~40 ha in Ku-ring-gai Chase National Park (Pittwater LGA) and ~30 ha in Bouddi National Park (Gosford LGA), as well as several small Council reserves in Pittwater and Gosford LGAs. Despite this increase in the conserved extent of this EEC, most of the distribution in both LGAs occurs along the reserve interface with urban lands, and is subsequently subjected to frequent low-intensity fire from hazard reduction burns, unmitigated clearing and weed invasion. Such ongoing threats dictate that PSGF remain listed as endangered in New South Wales.

Areas within Gosford and Pittwater LGAs supporting candidate-PSGF comprise very similar environmental characteristics. Both locations occur on shale-derived soils from Narrabeen series geology, both lie on the shores and associated low hills of Broken Bay and Pittwater, and both experience similar climate and rainfall. Further north around Lake Macquarie, a potentially different but clearly related form of forest is present on older Permian sediments (unpubl.

Table 3 Extant and estimated % loss figures for candidate–Pittwater Spotted Gum Forest (figures exclude ~40 ha mapped for Ku-ring-gai Chase NP by Thomas & Benson 1985).

LGA	Group	Extant (ha)	Pre-1750 (ha)	Loss (ha)	% loss
Pittwater	A (dry)	43	363	320	88
	B (moist)	69	385	316	82
Sub-total		112	748	636	85
Gosford	A (dry)	115	206	91	44
	B (moist)	-	-	-	-
Sub-total		115	206	91	44
Overall Total		227	954	727	76

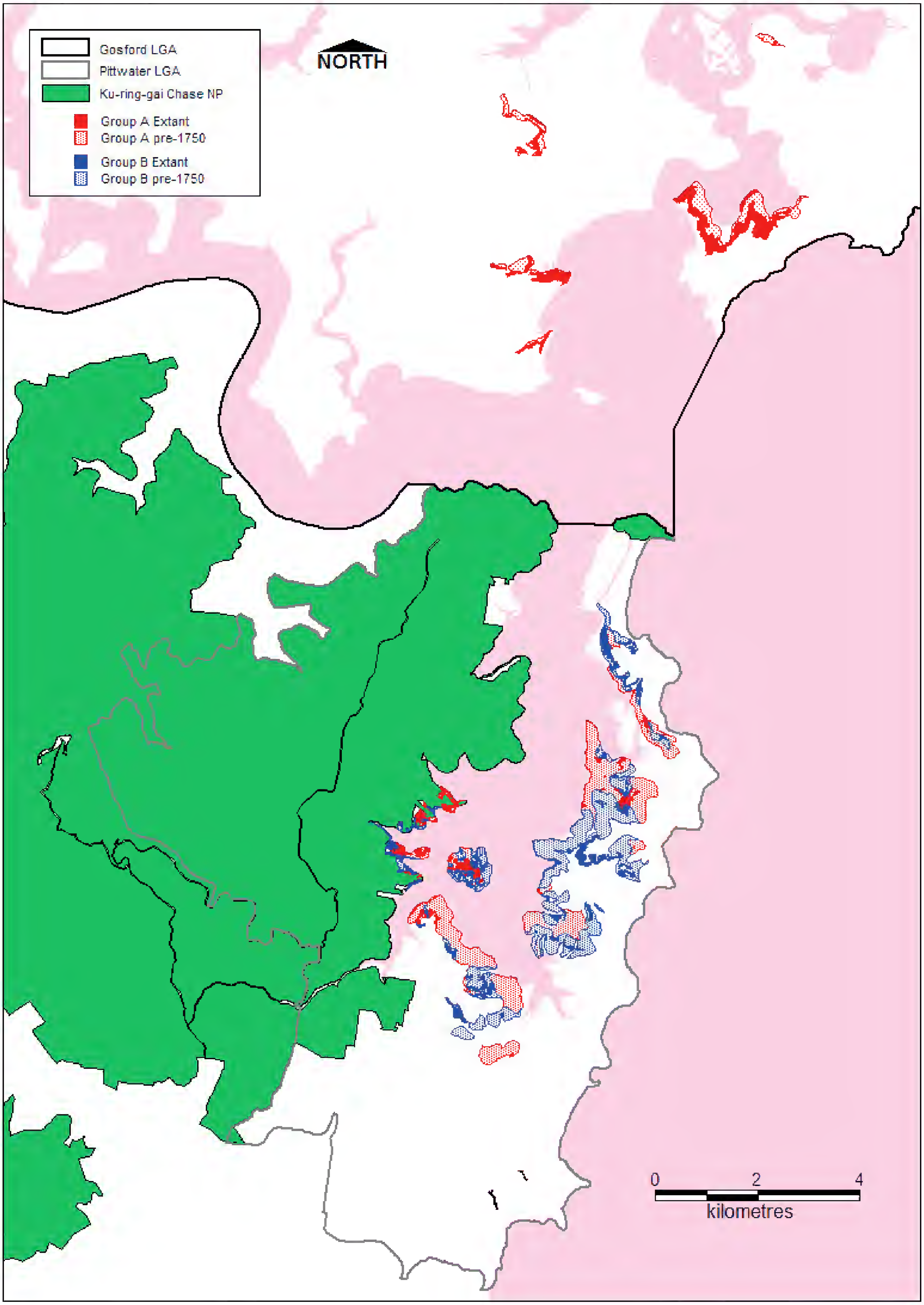

Fig. 5. Extant and pre-1750 distribution of candidate-Pittwater Spotted Gum Forest, here defined (excludes ~40 ha mapped for Ku-ring-gai Chase NP by Thomas & Benson 1985). Note that under the current determination, both extant and pre-1750 stands of PSGF within Pittwater LGA are considered part of the EEC.

data). To the south, as geology changes from Narrabeen to Hawkesbury outcropping south from Sydney, further stands of related vegetation do not reappear until the Nowra district where Narrabeen sediments again outcrop. Indeed, NSWDECCW (2010) did not define any communities dominated by *Corymbia maculata* along the coastline of the Sydney metropolitan area, but they were included in the work of Gellie (2005) for the southern forests.

Floristic variation within candidate-PSGF appears driven by aspect and associated soil moisture, and fire. All sites sampled within the Gosford LGA comprised the dry form of this community, while data from Pittwater LGA aligned with either the dry or moist form. Confounding the issue in the Pittwater LGA are extensive areas of littoral rainforest vegetation which, floristically, share many species with the moist form of PSGF (Bangalay & Eastcoast Flora Survey 2011). These linear patches of rainforest occur in the most protected gullies on sheltered aspects, and in many areas it is difficult to ascertain where the boundary between PSGF and littoral rainforest occurs. Absence of regular fire may be exacerbating this issue, allowing mesic species (eg: *Pittosporum undulatum, Glochidion ferdinandi*) to become established across a wider area (Rose & Fairweather 1997; Howell 2003).

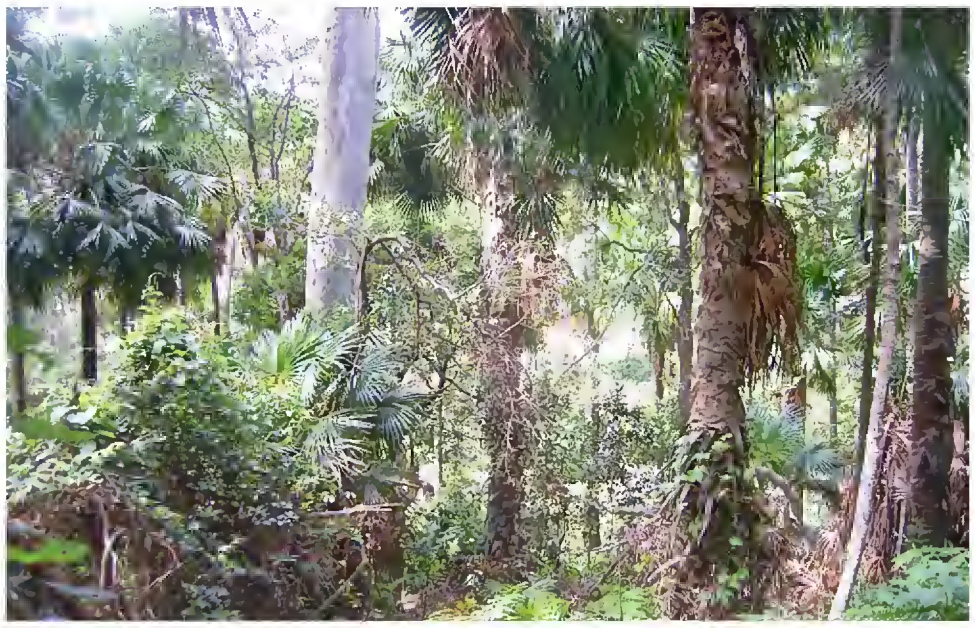

Fig. 6. Pittwater Spotted Gum Forest (Group B, moist form) from Palmgrove Park, North Avalon (Pittwater LGA). Note the co-dominance of *Livistona australis* beneath emergent *Corymbia maculata* and *Eucalyptus paniculata.*

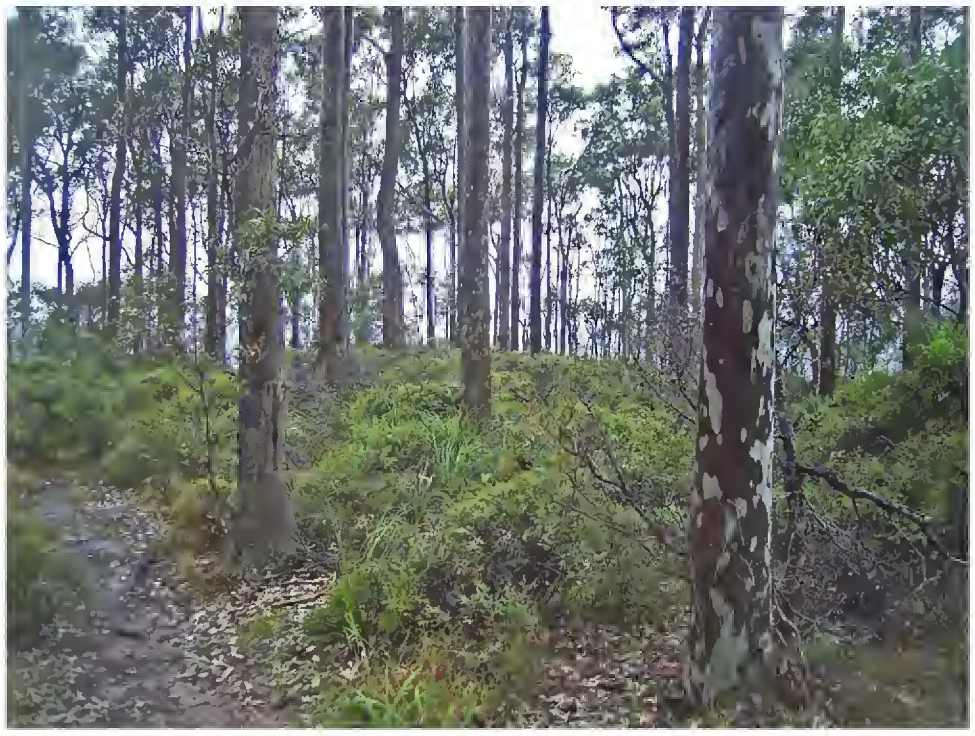

Fig. 7. Pittwater Spotted Gum Forest (Group A, dry form) from Woody Point (Pittwater LGA). At this location, *Acrotricha divaricata* dominates the shrub layer, but elsewhere this species is uncommon.

In an earlier study of Pittwater Spotted Gum Forest, Holden (2001) estimated there to be 51 ha from an original 345 ha of PSGF remaining within the Pittwater LGA, representing an 85% loss. Mapping completed to a higher resolution in the present study shows over double the extant and pre-1750 figures for the LGA, but retaining an 85% loss figure for Pittwater. For Gosford LGA, 44% of the original 206 ha has been lost to development, and combined the two LGAs show an overall loss of 76% of the original 954 ha. The principle reason for such a dramatic loss in PSGF has been clearing associated with rural and urban development, as the city of Sydney has expanded to accommodate an increasing population, particularly since the 1950s. Smith and Smith (1990) examined historical aerial photographs of the Barrenjoey Peninsula in Pittwater LGA, and concluded that between 1946 and 1986 forested lands declined from 47% to 8%, coinciding with increasing suburban development. Much of the cleared lands would have once supported PSGF. It is inevitable that native vegetation is deleteriously affected through population growth, as has occurred elsewhere in New South Wales (Benson 1987; Benson & Howell 1990b). Additionally, at most remaining stands, weed invasion (eg: *Lantana camara*, *Ligustrum sinense*) and fire exclusion are seen as the major threats to the ongoing survival of this community.

The existing determination for PSGF details 44 species which characterize the community (NSW Scientific Committee 1999). Using data from the present study, it is evident that 84% of the listed species are present within the dry form (Group A) of PSGF defined in this study, but only 48% of these occur in the moist form (Group B). This may be explained by the fact that the original determination presented a composite list of species for the community as a whole, rather than one tailoring the two forms. A revision of the determination may be warranted with this new information on the dry and moist forms of the community. In addition, clarification of the differences between the moist form of PSGF and areas of Littoral Rainforest EEC in the Pittwater LGA would be beneficial.

This study has examined the floristic composition of vegetation dominated by *Corymbia maculata* and *Eucalyptus paniculata* from two neighbouring LGAs on the northern outskirts of Sydney, showing them to be analogous. However, given the occurrence of potentially similar vegetation to the north in Lake Macquarie, and to the south in the Illawarra, Nowra and Batemans Bay, it is perhaps now desirable to further examine the floristic relationships existing between all of these areas. Recognition of the *Corymbia maculata-Eucalyptus paniculata* association within New South Wales by Pidgeon (1937) over 70 years ago should, at the very least, warrant more investigation of potential similarities, as all areas are seemingly subject to similar levels of threat.

Acknowledgements

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Changes in the area of coastal marsh in Victoria since the mid 19th century

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Abstract: European settlement in Australia has always been concentrated along or close to the coast. As a consequence, saltmarshes, mangroves and other coastal marshes have experienced a long history of modification and destruction. Depletion statistics are available for many coastal marshes in the Northern Hemisphere and, in Australia, for parts of New South Wales and Queensland. There are no equivalent State-wide data for Victoria. Using a suite of historical information, including extensive use of early surveyors' maps, we aimed to provide a consistent view of the change in the extent of coastal marshes since European colonization in Victoria (i.e. the mid-19th century). Notwithstanding the difficulties of interpretation, we estimate that prior to European colonization Victoria supported approximately 346–421 km^2 of coastal marsh, of which approximately 80–95% remains. Although a simplistic interpretation suggests a net loss of 5–20% in wetland area over this time period, it is clear that some parts of the coast have experienced relatively little change since the mid 19th century whereas others have been severely depleted and, in a few sectors, there may have been an expansion of coastal marsh. The situation with the Gippsland Lakes is complex, and according to the method used to interpret the original data sources there has either been a substantial increase or a loss of up to 35% in wetland area around Lake Wellington. The largest absolute losses have probably been of EVC 140 Mangrove Shrubland and of coastal saltmarsh dominated by *Tecticornia* spp. Parts of the coast where significant losses have occurred include the Lonsdale Lakes, western shore of Port Phillip Bay, Anderson Inlet, Shallow Inlet, Powlett-Kilcunda, Corner Inlet and Nooramunga, and possibly Lake Wellington. With the exception of the Lonsdale Lakes, all these areas are situated along the Gippsland coast. Changes to coastal marshes have not stopped and are unlikely to cease in the near future. The destruction of coastal marshes for industrial development remains an ongoing threat in many regions (e.g. in Western Port) and is likely to be compounded by climate change and, in particular, sea-level rise.

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Introduction

Because of their position along the coast where land is acquired for human settlement, saltmarshes, mangroves and other coastal wetlands have experienced a long history of modification and destruction. Gedan *et al*. (2009, page 117) concluded that 'since the Middle Ages, humans have manipulated saltmarshes at a grand scale, altering species composition, distribution, and ecosystem function'. In some cases, such as across much of northern Europe, coastal saltmarshes have been used for agricultural purposes since at least the 17th century and possibly even as early as the 13th century (Adam 1990, French 1997). In other cases, it is known that physiographic changes to estuaries since pre-Roman times have had marked impacts on the distribution of coastal marshes (Steers 1977). The European experience was broadly repeated in North America: Gedan & Silliman (2009) estimated that approximately 40% of coastal saltmarsh has been lost from the North Atlantic coast and over 90% from the Pacific coast of North America since European colonization (baseline dates range from 1609 to 1893). The greatest early losses probably arose as a result of the direct and wholescale conversion of marshes for agricultural or urban uses or for the salt industry, whereas over recent decades a suite of other indirect factors have become responsible, including subtle alterations to wetland hydrology and nutrient loading, as well as the introduction of introduced species, both plant and animal (e.g. see Morris *et al*. 2004, Silliman *et al*. 2005, Jefferies *et al*. 2006).

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Because of the country's more recent colonization by Europeans, Australian coastal marshes do not share the centuries-old history of large-scale modification that characterizes so many Northern Hemisphere systems. Nevertheless, almost all coastal areas of south-eastern Australia have experienced some degree of alienation and habitat modification or destruction over the past 150–200 years. In the Koo Wee Rup Swamp on the northern shore of Western Port (Victoria), for example, large areas of open wetland and paperbark swamp (*Melaleuca ericifolia*) were 'reclaimed' for agriculture in the late 19th and early 20th century, as described enthusiastically by East (1935) and by Roberts (1985); see Yugovic & Mitchell (2006) for a detailed analysis of changes in this region since European colonization. Similarly, Bird (1980*a*, 1980*b*) reported that, in places, the mangrove fringe of Western Port had been reduced markedly by clearing, infilling, drainage and die-back. Changes to the shoreline shown in surveys of the mid 19th century were evident by 1939 (Bird 1980*a*, 1980*b*) and have continued apace since then. A boat channel, for example, was cut into mangroves and saltmarsh in 1967 at Yaringa (Bird 1971), and even more extensive port, marine and industrial developments have taken place over the past two decades and, indeed, are reputed to be planned for the near future for this mangrove- and saltmarsh-fringed coastal embayment.

In the late 19th Century, Victorian mangroves and coastal saltmarsh plants were heavily exploited for their soda-ash content (Bird 1978, 1981). Since tallow was abundant (because of the large number of sheep) but soap relatively scarce, many types of native vegetation were burnt to yield the ash needed for soap-making (Adam 1990). Halophytic plants were an excellent source of ash, and Carr & Carr (1981) and Whinray (1981) noted that *Tecticornia arbuscula*, *Sarcocornia quinqueflora*, *Atriplex cinerea* and *Rhagodia candolleana* (the latter being a common coastal species which is rare and only opportunistic in saltmarshes: Geoff Carr, pers. comm) were used in the same way. Mangroves, however, were the main source of ash, and their utilization to generate barilla was so intense that Bird (1978, 1981) argued that clearing mangroves for ash production has had long-term impacts on their distribution in Victoria. Indeed, in his review of the natural history of French Island (in Western Port), Lacey (2008) noted that the first Europeans recorded as living on the island (~1843/1844) were engaged in burning mangroves to produce barilla for soap-making.

Large expanses of coastal saltmarsh were also alienated for salt production in the 19th century between Melbourne and Geelong, on the western shores of Port Phillip Bay and Corio Bay, where as a result of low rainfall (~600 mm per year) evaporation greatly exceeds precipitation. Shell-grit mining has been undertaken in other coastal saltmarshes around Victoria, mostly to produce lime, for road building, and as source of calcium carbonate for the poultry industry. Much of this mining has occurred in areas adjacent to or supporting saltmarsh, predominantly on the western shores of Port Phillip Bay and Corio Bay from Melbourne to Point Lonsdale, but also in the Point Lonsdale–Collandina area. Moreover, Laegdsgaard (2006) noted that many Australian saltmarshes, especially in New South Wales, were on private land and subject to stock grazing. This is certainly the case also for Victoria, where coastal saltmarsh is routinely grazed by cattle (particularly in western Victoria and Gippsland) and sheep (e.g. around Port Phillip Bay), as well as being grazed by introduced animals such as feral deer and goats (e.g. on French Island).

Finally, the exotic and invasive *Spartina* (*Spartina anglica* and *Spartina* x *townsendii*) is present in a number of Victorian saltmarshes and other coastal areas. The first documented introduction of *Spartina* to Australia was a planting in Corner Inlet, probably in the 1920s, by Professor AJ Ewart from the University of Melbourne (Boston 1981). Williamson (1996) provided an overview of *Spartina* introductions into Victoria. As reported by Bird & Boston (1968), early plantings of *Spartina* occurred at Lake Connewarre, the Bass River, Corio Bay, Anderson Inlet, and several locations in Corner Inlet in the 1920s and 1930s. Although most of the original introductions were unsuccessful, in some parts of Victoria *Spartina* did expand rapidly. The spread of *Spartina* through Anderson Inlet, for example, started to be noticed in the 1980s and the Department of Conservation and Natural Resources began to investigate it in 1991 (Blood 1996, Williamson 1996). Mapping in 1993 showed there were 150–280 ha of *Spartina* in Victoria, with most in Anderson Inlet (108 ha confirmed, 130 ha suspected), Corner Inlet (42 ha) and small infestations in Shallow Inlet, Western Port and the Barwon River (Williamson 1996).

Saintilan & Williams (2000) reviewed the records of saltmarsh loss in eastern Australia and, based on 28 surveys employing historical aerial photographs, showed that there had been a widespread loss since the 1940s–1950s. Reported losses were as high as 100% for Weeney Bay (part of Botany Bay, New South Wales, from 1950–1994) and 67% for the Hunter River estuary (excluding Hexham) from 1954–1994. Harty & Cheng (2003) reported a loss of 78% of saltmarshes in Brisbane Water, near Gosford, New South Wales, between 1954 and 1995. The studies reviewed by Saintilan & Williams (2000) were overwhelmingly taken from Queensland and New South Wales, and the sole Victorian study was that by Vanderzee (1988), who reported extensive – but unquantified – loss of coastal saltmarsh in the Corner Inlet region.

Ross (2000) reviewed changes to the distribution of mangrove and coastal saltmarsh around parts of Western Port, using the 1842 maps of the surveyor George Douglas Smythe as a baseline. She found evidence of clearing of mangroves in particular (for boat access and to burn for barilla production) but did not make quantitative estimates as to the magnitude of wetland loss or expansion. In one of the few quantitative studies undertaken of the Victorian coast, Ghent (2004) compared past and present distributions of coastal saltmarsh in Port Phillip Bay, and concluded that ~65% of pre-European saltmarsh had been lost, mostly before 1978.

Saintilan & Rogers (2001) surveyed mangroves and coastal saltmarsh around Quail Island (Western Port) and found an increase of 32% over the period 1973–1999. Gullan (2008) estimated that about 30% of Victorian coastal saltmarsh had been permanently cleared for coastal or marine development.

It is the case, therefore, that we have little or no quantitative understanding of how coastal marshes across Victoria as a whole have changed since European colonization. This situation is partly the result of regional studies employing different methods or concentrating on areas of different scale. State-wide figures cannot be extrapolated from isolated studies of given localities, as different places with different histories may have experienced locally large losses, expansions or remarkable stability. These local differences relate to idiosyncratic site-specific histories, and may involve natural geomorphological changes or human-induced changes of many kinds.

The current investigation aims to provide a consistent view of the change in the extent of coastal marshes in Victoria since European colonization (i.e. the mid-19th century).

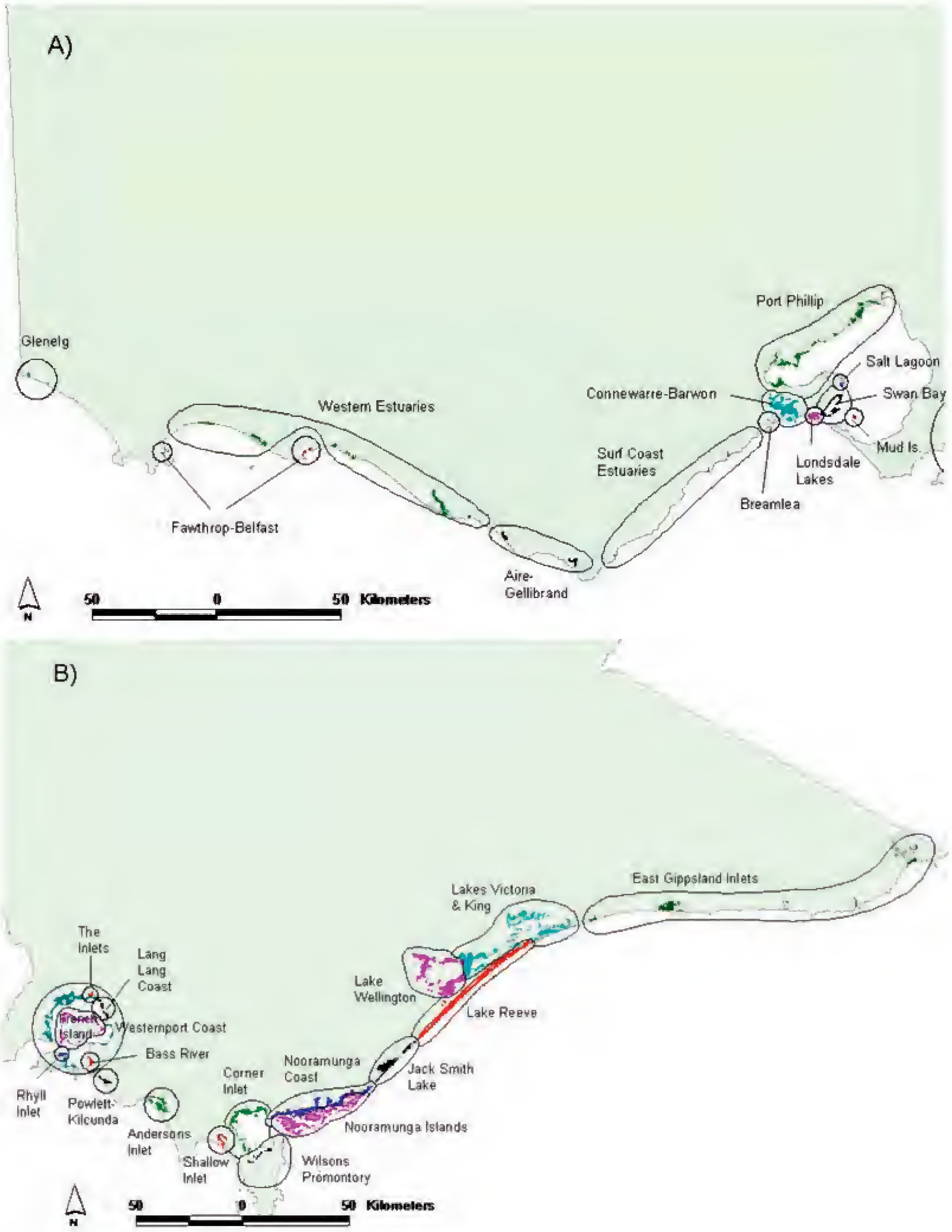

Fig. 1. Division of the Victorian coast into sectors for the purposes of calculting losses or gains in each region. A) shows western and central Victoria; B) shows eastern Victoria.

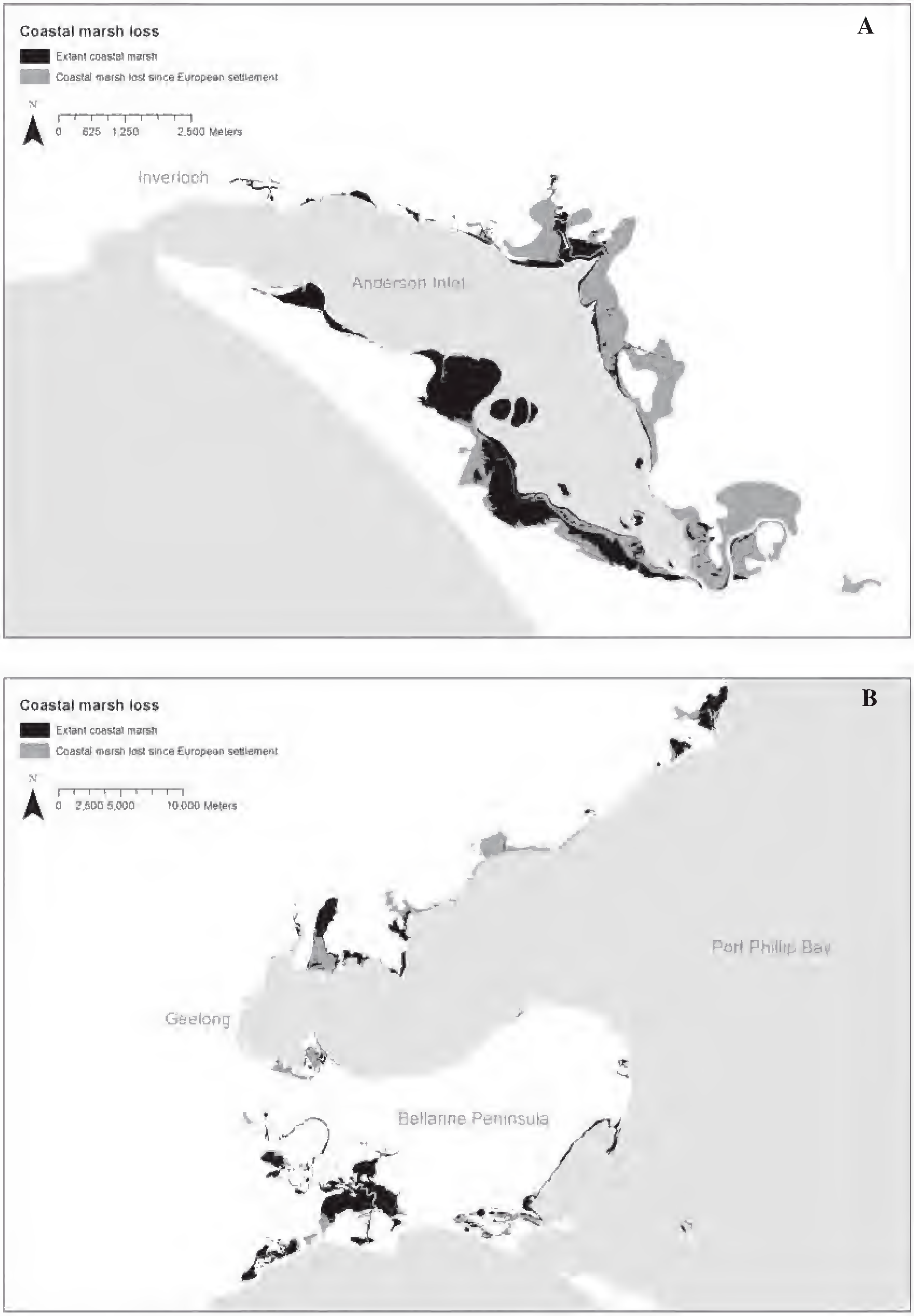

Fig. 2. Examples of the extent of loss of coastal marsh in Victoria. A) shows Anderson Inlet; B) shows the western shore of Port Phillip Bay and the Bellarine Peninsula. The black shading shows current marsh distributions; middle grey shading shows areas of marsh lost since European colonization; light grey shadings are open water; white is terrestrial land.

It elaborates on work undertaken as part of the Victorian Saltmarsh Study (2011), which showed that the present-day area of coastal saltmarsh and estuarine wetland (saline coastal wetlands dominated by halophytic shrubs, herbs, grasses or sedges) in Victoria was 224 km^2 and that of mangroves was 52 km^2. The specific aim of our investigation was to quantify the loss of coastal marshes, including mangroves, saltmarsh and estuarine wetland, since European colonization.

Methods

Information sources

We used three approaches to quantify changes in the area of coastal wetlands across Victoria since European colonization. The first was to interrogate coastal maps prepared in the mid 19th century, mostly by surveyors and geologists. In some places, early surveyors' plans show saltmarsh (often referred to as 'samphire') and mangrove patches quite clearly. In other cases, low water, high water and the spring-tide maximum are shown as lines. On some plans, wetlands are shown and labelled 'salt' or 'fresh', thus allowing some assumptions to be made as to the plant assemblages or wetland types that formerly occupied the site. Other maps show unannotated wetlands that are ambiguous, in which case we used current-day information to infer the likely wetland type. We searched for useful early maps using the catalogues of the National Library of Australia and the State Library of Victoria, along with sections of the Public Records Office microfiche collection. Sources are shown in Table 1.

Second, we used on-ground interpretation. As part of the preparation of the State-wide saltmarsh inventory, we (and others) undertook extensive ground-truthing of oblique aerial photographs (Victorian Saltmarsh Study 2011). Where field teams noted evidence of wetland infilling or modification, such evidence was recorded and, where possible, the likely former extent of the wetland marked on the aerial images. Although this was sometimes a simple task, the interpretation is nevertheless often confusing, particularly where no vestige of the former high-tide boundary remained or where obvious palaeo-coastal features from periods much earlier than European colonization (e.g. from the Holocene maximum: see Pirazzoli & Pluet 1991) could be mistaken for more recent shorelines.

Third, we used remote-interpretation tools directly. Where there were no useful early maps or sites were not visited in the field, aerial photographs were re-viewed to check for clues about the prior vegetation and wetland extent. Evidence of channels or levees used to drain low-lying coastal plains, for example, was taken as evidence of former saltmarsh vegetation. We also used elevation data to check that our view of pre-European saltmarsh only included low-lying areas. As we did not have access to LiDAR elevation data at the time for the entire Victorian coast, we used the freely-available elevation data from the NASA Endeavour 'Shuttle Radar Topography Mission' 2000 (see Farr *et al*. 2007). Remote-sensing tools were particularly helpful in the Corner Inlet and Nooramunga areas, where the historic record is not detailed and physical access is difficult. (Note that since this study was undertaken, LiDAR data have become available for the entire Victorian coastline and they should prove to be valuable for future investigations into wetland dynamics and sensitivity to anthropogenic change.)

Vegetation units and wetland types

In Victoria the system used for classifying and mapping native vegetation is based on Ecological Vegetation Classes (EVCs), which are defined as one or a number of floristic and structural types that appear to be associated with a recognizable environmental niche and can be characterized

Fig. 3. Flowerhead of Native Sea Lavender, *Limonium australe*, a saltmarsh plant listed as Vulnerable in Victoria.

Fig. 4. Creeping Brookweed, *Samolus repens*, a common herb in Victorian coastal saltmarsh.

Table 1. Summary of historic sources used to infer pre-European extent of coastal marshes in Vicotria. Maps are listed chronologically. When the map or source material is undated (or date illegible), we refer to the item as 'undated', unless there is an approximate or apparent date of production (e.g. in marginal annotations), in which case we use the abbreviation 'c.'. Where the material is untitled, we provide a brief description of the map in square brackets. The name of the public institution holding the map is abbreviated as follows: Public Records Office of Victoria: PROV; National Library of Australia: NLA. The microfiche sets of the Public Records Office of Victoria are abbreviated as follows: Gippsland Rivers: GR; Coastal Surveys: CS; Special Surveys: SS.

Date	**Surveyor**	**Title [or description]**	**Source**
1837	Russell R	Map shewing the site of Melbourne and the position of the huts & buildings previous to the foundation of the township by Sir Richard Bourke in 1837.	NLA MAP RM 1288
1840	Myers CI	Rough sketch of part of the district of Gipps Land based on Mr Ass. Surveyor Townsends' chain measurement from Port Albert to Omeo shewing the positions and area of the several stations.	PROV GR24
1841	Anon.	Survey of part of the Parish of Moolap, 1841, suburban lots.	PROV [location unknown]
1842	Anon.	Plan of the town of Alberton on the River Albert.	PROV S3A
1842	Smythe GD (a)	Sheet No. 5 of Western Port Coast	PROV CS69
1842	Smythe GD (b)	Survey of the islands of Western Port	PROV CS16
1842	Smythe GD (c)	Survey of the eastern coast of Western Port Sheet No 6.	PROV CS15A
1842	Smythe GD (d)	Survey of the N west coast of Western Port from Toorndeeck to Baangring	PROV [location unknown]
c.1842	Anon.	[untitled. Map of Alberton area]	PROV SS1
1843	Anon.	Survey of the proposed town reserve at Port Fairy.	PROV CS37A
1843	Hoddle R	Survey of part of the Parish of Paywit County of Grant.	PROV SP18
1843	Myers CI	Plan of Port Fairy shewing the position of Mr Atkinson's special survey.	PROV SS8
1843	Smythe GD	special surveys at Port Albert and some portions for sale on the River Tarra.	PROV CS4
c.1843	Anon.	[untitled. Map of Port Fairy area]	PROV SS8
1847	Smythe GD	Survey of the rivers and creeks from the Barwon Heads to Point Roadknight.	PROV CS30B
c.1847	Smythe GD	Survey of the coast from Cape Patterson to Cape Liptrap with rivers, creeks, lakes, marshes, swamps, scrubs, plains and ranges within one days work of coast.	PROV CS14
1848	Smythe GD	Plan of part of the Gipps Land District from Cape Patterson to Shallow Inlet and from Cape Wellington to Shallow Inlet West.	PROV CS43
1849	Wright	Plan of Corner Inlet Gippsland shhewing the town of Alberton.	PROV CS52
1849	Wilkinson J	Survey of the northern shore of Lake Wellington, The Straits, Tom's Creek etc.	PROV CS51
1849	Wilkinson J	Survey of the Nicholson and part of the Tambo Rivers.	PROV GR9
c1849	Wilkinson J	Survey of the Macarthur or Mitchell River from Lake King to the ford at Lindenow and Lake Victoria from Jones' Backwater to Storm Point.	PROV GR6A
1850	Anon.	Plan of the Town of Belfast, Port Fairy.	PROV SB3
1850	Garrard & Shaw	Map of the town and suburbs of Geelong comprising the lands in the parishes of Gheringhap, Moorpanyal, Barrabool, Duneed, Moolap, Bellerine & Paywit: together with the positions of the bar and the proposed improvements.	NLA MAP RM 1963
1853	Barrow J	Survey of Lady Bay, Warrnambool.	PROV CS35B
1853	Christie	[untitled. Map of Melbourne]	NLA MAP RM 3473
1854	Barrow J	Warrnambool shewing a plan to prevent the accumulation of sand in Lady Bay.	PROV CS98
1855	Byerley J	Subdivision of sections in the parish of Connewarre County of Grant Part 4.	NLA MAP NK 2456/217
1855	Lees EH	Plans of Wingan River, travserse from inlet northwards, County of Croajingalong.	PROV GR53
1855	Watson	Country lands in the parishes of Codrington and Eumerella, Counties of Normanby and Villiers.	NLA MAP NK 2456/216
1859	Ross R	Corio Harbour, Geelong Bay	PROV CS29A
1862	Daintree R & Ross JL (a)	Geological Survey of Victoria No. 29.	NLA MAp RM 2335/29 Tile 1a
1862	Daintree R & Ross JL (b)	Geological Survey of Victoria No. 32.	NLA MAP RM 2335/23 Tile 1b
1863	Daintree R	Geological Survey of Victoria No.23	NLA MAP RM 2335/23 Tile a1
1863	Daintree R & Shepherd R (a)	Geological Survey of Victoria No. 23..	NLA MAP RM 2335/23 Tile b2
1863	Daintree R & Shepherd R (a)	Geological Survey of Victoria No. 29.	NLA MAP RM 2335/29 Tile a2
1863	Daintree R, Wilkinson CS & Shepherd R	Geological Survey of Victoria No. 29	NLA MAP RM 2335/29 Tile a3

1864	Cox HL	Port Phillip	NLA MAP RM 1696
1866	Bailliere FF	County of Mornington.	NLA MAp RaA 16 (Pl. 13)
1869	Stonley HJ (a)	Portland Bay	PROV CS38
1869	Stanley HJ (b)	Coast from Cape Wollamai to Cape Liptrap	PROV CS14
1869	Stonley HJ (c)	Venus Bay and Anderson's Inlet	PROV CS62A
1874	Cox HL & Stanley HJ	Entrance to Port Phillip including the banks and channels.	NLA MAP RM 2379
1880	Noone J	Cocooc, County of Grant	NLA MAP RM 2741/287
1880	Noone J	Murtcaim, County of Grant	NLA MAP RM 2741/310
1896	Goldsmith L & McGauran TF	Deutgam, County of Bourke	NLA MAP RM 3013
undated	Anon. (a)	[untitled.Map of Andersons Inlet and Cape Liptrap	PROV CS43D
undated	Anon. (b)	[untitled. Map of Alberton area]	PROV CS55
undated	Anon. (c)	[untitled. Map of Alberton area]	PROV SS1
undated	Anon. (d)	Plan of survey of sea coast from Lake Reeve to Snowy River with part of Lake Victoria	PROV CS3
undated	Anon. (e)	[untitled. Map of Gippsland Lakes]	PROV CS42
undated	Anon. (f)	Snowy River Coastal Survey 34	PROV GR32
undated	Barrow J	Plan shewing the mouth of the River Moyne at Belfast	PROV CS97
undated	Proeschel F	Map of Melbourne and suburbs	NLA MAP RM F878
undated	Smythe GD (a)	Survey of the coast from Cape Liptrap to Shoal Inlet and one day works inland with the line to Corner Inlet, Mr Townsend's Survey.	PROV CS14C
undated	Smyhe GD (b)	Survey of Shoal Inlet and Part of the Ninety Mile Beach to Mr Surveyor Townsends marked tree at the mouth of the Tambo River.	PROV CS8
undated	Smythe GD (c)	Plan of coast survey from Stas. C to D. Part of Lake Victoria, Lake Reeve, Raymond Island, Lake King to the mouth of the Mitchell and Tambo Rivers.	PROV CS4 & CS5
undated	Smythe GD (d)	Survey of part of the Ninety Mile Beach with rivers, creeks, lakes, marshes, roads and stations within one days work of the sea-coast.	PROV CS4 & CS5
undated	Smythe GD (e)	Survey of Coast from Merriman's Creek to Lake Reeve	PROV CS6
undated	Urquart WS	Rough sketch from- plan of the Merri River shewing the situation of the last special survey selected by Mr Wm Rutledge.	PROV SS6B

by their adaptive responses to ecological processes that operate at the landscape scale. The EVC approach to vegetation classification therefore uses floristic and structural criteria, combined with geographic information on niches and distributions (Oates & Taranto 2001, Department of Natural Resources and Environment 2002).

The different methods for determining pre-European vegetation patterns varied in their ability to resolve different vegetation and wetland types, but generally were poor. Because of this limitation, we chose to represent only two mapping units:

- Mangroves. All vegetation anywhere in Victoria where *Avicennia marina* occurred, as per EVC 140 Mangrove Shrubland (Department of Sustainability and Environment 2009).
- Other Coastal Marsh. This unit includes all intertidal wetlands not dominated by *Avicennia marina*. For the purposes of our analysis, it includes saltmarshes dominated by succulent chenopods, sedges (e.g. *Gahnia trifida*), rushes (*Juncus kraussii*), grasses (e.g. *Distichlis distichophylla*) and the herb *Wilsonia* spp. (Some of the dominant species occur widely near the coast, so only those areas that were sometimes inundated by saline water were included). In many estuaries it is difficult to determine where tidally inundated areas meet river waters. As a simple solution, we included all estuarine marshy areas currently dominated by *Juncus kraussii* and excluded all areas dominated by *Phragmites*, *Typha*, *Cladium*, *Bolboschoenus*, *Schoenoplectus* or any woody species. Brine pools vegetated by plants that grow ephemerally after inundation (e.g. *Ruppia* spp., *Lepilaena* spp.) were included if they were surrounded by coastal saltmarsh. Given the above interpretation, our circumscription corresponds to EVC 9 Coastal Saltmarsh Aggregate, EVC 10 Estuarine Wetland, EVC 196 Seasonally Inundated Sub-saline Herbland and, where occurring within a marsh, EVC 842 Saline Aquatic Meadow and bare ground.

To reflect the uncertainty about which vegetation types grew where, the general term 'coastal marsh' is used below to refer to relevant units where their type is not precisely known. Table 2 shows a summary of the different vegetation units, classified according to EVCs.

Division of coast into sectors

For the purposes of description and reporting, we divided the coast into 30 sectors that represent discreet areas with relatively homogeneous geomorphology, rainfall and land-use. They are not all of the same size, and some contain much more wetland than others. Whilst most sectors are nested within a single bioregion, some straddle two (e.g. Breamlea). The sectors are shown in Figure 1 and briefly described in Table 3.

Data manipulation and map generation

The information from the diverse information sources was used to compile a map of the likely pre-European distribution of relevant coastal wetlands across Victoria within each of the 30 coastal sectors. The distribution of wetlands was prepared as a digital map dataset as a series of spatially referenced polygons. All polygons were drawn by hand, using Arcview 3.2 (ESRI), and traced directly from historical sources or current-day aerial photographs. A stream digitiser was used with a sensitive screen and stylus (Minnesota Department of Natural Resources 2000; using a Wacom tablet screen) to allow fluid outlines to be drawn. This instrumentation allowed direct tracing of the shapes on the screen in the same manner as has been used traditionally with a pencil and tracing paper over an annotated map image. The polygons were attributed with a series of fields describing their vegetation type, etc.

In cases where the early cartography was good and vegetation patterns relatively simple, polygons were captured using transparencies traced from the historic plan placed over the screen. Where the original plans were distorted in relation to our maps, for example due to cartographic errors or different projections, the historic plans were captured digitally and geo-rectified to align in space with our map data. This was done by digitally stretching the images against known reference points, using the 'ImageWarp' software available as an extension to Arcview 3.2 (ImageWarp version 2.0, 1999). The rectified images were then used as templates in the same way as the aerial photographs. In cases where the information was insufficient to geo-rectify the image or where the level of accuracy in the original product was very poor, we interpreted the historic information 'by eye'. In

Table 2: Range of estuarine vegetation types referred to in the study, classified by Ecological Vegetation Class (EVC).

EVC	EVC name	Characterization	Indicator species
10	Estuarine Wetland	Rushland/sedgeland vegetation, variously with component of small halophytic herbs, occurring in coastal areas where freshwater flows augment otherwise saline environments.	*Juncus kraussii*, occasionally with *Phragmites australis* or species of Cyperaceae.
842	Saline Aquatic Meadow	Submerged ephemeral or perennial herbland of slender monocots, occurring in brackish to saline water bodies subject or not to dry periods. The vegetation is characteristically extremely species-poor, consisting of one or more species of *Lepilaena* and/or *Ruppia*.	Variously *Ruppia megacarpa*, *Ruppia polycarpa*, *Lepilaena* spp. (e.g. *L. preissii*, *L. bilocularis*, *L. cylindrocarpa*).
140	Mangrove Shrubland	Extremely species-poor shrubland vegetation of inter-tidal zone, dominated by mangroves.	Characteristically occurs as mono-specific stands of *Avicennia marina*. In some stands, species from adjacent Coastal Saltmarsh or Seagrass Meadow also present.
196	Seasonally Inundated Sub-saline Herbland	Very species-poor low herbland of seasonal saline wetland within relicts of former tidal lagoons, dominated by *Wilsonia* spp.	*Wilsonia humilis* sometimes with *W. backhousei* and/or *W. rotundifolia*.
952	Estuarine Reedbed	Vegetation dominated by tall reeds (usually 2-3 m or more in height), in association with a sparse ground-layer of salt tolerant herbs. Distinguished from Estuarine Wetland by the vigour and total dominance of reeds, and from Tall Marsh by the presence of halophytes.	*Phragmites australis*, with associated species variously including *Samolus repens*, *Juncus kraussii*, *Triglochin striatum*, *Bolboschoenus caldwellii* and *Suaeda australis*.
953	Estuarine Scrub	Shrubland to scrub of myrtaceous shrub species of sub-saline habitat, occurring in association with ground-layer including halophytic herbs.	*Melaleuca ericifolia* (in eastern Victoria), with other *Melaleuca* spp. (e.g. *Melaleuca lanceolata*, *Melaleuca gibbosa*) or *Leptospermum lanigerum* in marginal sites in western Victoria. Gound-layer includes *Samolus repens*, *Triglochin striata* and *Selliera radicans*, variously with *Sarcocornia quinqueflora*, *Gahnia filum*, *Poa poiformis*, *Juncus kraussii*, *Disphyma crassifolium*, *Distichlis distichophylla*.

Table 3: Division of the Victorian coast into sectors for the purposes of calculting losses or gains in each region. Rainfall is approximate (to the nearest 100 mm) and refers to the immediate area not the entire catchment. The Bioregion names are abbreviated as per Department of Sustainability and Environment: Brid (Bridgewater); VVP (Victorian Volcanic Plain); WaP (Warrnambool Plain); OtP (Otway Plain); GipP (Gippsland Plain); WPro (Wilsons Promontory) and EGL (East Gippsland Lowlands).

Sector	**Geomorphic context**	**Major adjacent land-uses**	**Bioregion**	**Rainfall (mm)**
Glenelg	Estuary, limestone and calcareous barrier system.	Native vegetation, Agriculture, Plantation	Brid	800
Fawthrop-Belfast	Estuaries, lagoons with prominent flood-tide deltas, channelized openings, basaltic lowlands and barrier and dune systems.	Agricultural, Urban	VVP WaP	700
Western estuaries	Estuaries, subdued hinterland, barrier and dune systems.	Agricultural	WaP	700
Aire-Gellibrand	Large estuarine floodplains, steep hinterland and barrier systems.	Native vegetation Agricultural	OtP WaP	1000
Surf-coast estuaries	Small coastal floodplains, barrier systems, steep hinterland.	Native vegetation Agriculture, Urban	OtP	700
Breamlea	Estuary with barrier system, sand and silt deposits adjoining basalt.	Agricultural	OtP (VVP)	600
Connewarre-Barwon	Large estuarine floodplain, prominent flood-tide delta, sand and silt deposits, barrier spit, surrounding geology diverse.	Agricultural, Urban	OtP (VVP)	600
Lonsdale Lakes	Saline lakes isolated from ocean by complex sand and silt deposits, limestone.	Agricultural , Urban	OtP	700
Salt Lagoon	Saline lake isolated from the ocean.	Agricultural , Urban	OtP	700
Swan Bay	Embayment, sheltered by series of sand spits.	Agricultural , Urban	OtP	700
Mud Islands	Sand and shell-grit shoals, anchored on calcarenite.	None / Ocean Native vegetation	NA	700
Port Phillip	Coast of large embayment, largely low-relief basaltic hinterland.	Water treatment Agricultural, Urban Industrial	VVP OtP	600
The Inlets	Partly channelized outflow of drained swamp, tidal channels from sheltered coast of embayment	Agricultural	GipP	800
Western Port	Low-energy coast of large embayment, minor estuaries, large tidal range, extensive mudflats.	Agricultural, Urban Native vegetation	GipP	800
French Island	Coast of large island within Western Port, mostly sandy deposits, large tidal range.	Native vegetation Agricultural	GipP	800
Rhyll Inlet	Sheltered embayment behind sand spit.	Urban, Agricultural	GipP	800
Lang Lang coast	Cliffed and eroding coast.	Agricultural	GipP	800
Bass River	Large floodplain on Western Port coast.	Agricultural	GipP	800
Powlett-Kilcunda	Small estuaries.	Agricultural	GipP	800
Anderson Inlet	Shallow, sheltered inlet behind barrier, estuarine input.	Agricultural, Native vegetation, Urban	GipP	800
Shallow Inlet	Shallow, sheltered tidal inlet behind barrier.	Agricultural	GipP	800
Wilsons Promontory	Sheltered portions of coast.	Native vegetation	WPro	900
Corner Inlet	Sheltered coast, geology various, minor estuaries, extensive sand- and mud-flats.	Agricultural, Urban Native vegetation	GipP	900
Nooramunga coast	Sheltered coast opposite Nooramunga Islands, geology various, minor estuaries, extensive sand- and mud-flats.	Agricultural, Urban Native vegetation	GipP	800
Nooramunga islands	Complex of quartz sand barrier deposits forming sheltered inlets, extensive sand- and mud-flats.	Native vegetation Agricultural	GipP	800
Jack Smith Lake	Lake separated from ocean by sand barrier.	Agricultural	GipP	700
Lake Reeve	Saline lake system separated from ocean by sand barriers. Bordered by complex system of contraction ridges.	Native vegetation Agricultural, Urban	GipP	700
Lake Wellington	Coastal lagoon separated from ocean by large sand barriers.	Agricultural Native vegetation	GipP	700
Lakes Victoria and King	Coastal lagoons separated from ocean by large sand barriers.	Agricultural, Urban Native vegetation	GipP	700
East Gippsland inlets	Inlets, barrier-built lagoons and estuaries.	Native vegetation	EGL	900

some cases the different pieces of evidence were at variance. Multiple historic plans, and the evidence visible on the aerial imagery do not always tell precisely the same story and in all such cases a subjective compromise was made.

Calculating depletion statistics

Calculating accurate depletion statistics is complicated by the different resolution of the current and 19th century maps. We excluded areas of bare ground, brine pools and other minor inlying patches of non-saltmarsh vegetation generated by the Victorian Saltmarsh Study (2011), as we could not predict the distribution of these inliers on a suitably fine scale in the pre-European distribution map. Thus, we compared the pre-European area of coastal marsh to an increased current saltmarsh total that included these inlying areas; even so, the inflation factor was, on average, only 1.1. In the absence of other information, we assumed also that the relative proportions of ECV 10 Estuarine Wetland and EVC 9 Coastal Saltmarsh Aggregate remained constant during any process of historical depletion. As our study was undertaken at a State-wide scale using historic plans of limited accuracy, we could not capture all the very small changes that had occurred. Some of the changes found in other local studies using different techniques would be barely detectable at our scale (e.g. Vanderzee 1988, Saintilan & Rogers 2001). Small changes to the sea-ward mangrove fringe are particularly difficult to detect from our sources.

Results

Table 4 shows the estimated area of mangroves and other coastal marshes before European colonization ('pre-1750 area') and the present-day area for various sectors of the Victorian coast. The degree of retention is shown also for the two marsh types in each of the sectors. Because the historic resources that were available generally provided little or no information about non-mangrove coastal marshes, it was necessary to combine all of the non-mangrove wetland types into one group, called Other Coastal Marsh (Table 4). In the original sources these diverse wetland types were often labelled with an imprecise array of labels, such as 'salt', 'samphire' or 'saltmarsh'.

The Gippsland Lakes area presented particular problems for calculating depletion statistics, as there have been potentially large gains or proportionally smaller losses along this section of the Victorian coast, especially for Lake Wellington. The primary difficulty is with existing areas of coastal saltmarsh, some of which are natural occurrences, some of which seem to be expansions of saltmarsh since European colonization. Given this uncertainty, we calculated upper and lower bounds on the depletion estimates for the Gippsland Lakes sector, based on two extreme scenarios:

> Scenario 1: All the ambiguous saltmarsh areas are natural.
>
> Scenario 2: All the ambiguous saltmarsh areas are recent expansions, in which case they are counted as gains which offset other losses.

Our analysis indicates that, in the mid 19th century, Victoria supported approximately 346–421 km^2 of coastal marsh, of which on a State-wide basis approximately 80–95% remains (depending on which of the two scenarios for the Gippsland Lakes are employed). The overall value of 80–95% can be further dividied into values of 92% remaining for mangroves on a State-wide basis, and 75–95% for the other types of coastal wetland (Table 4).

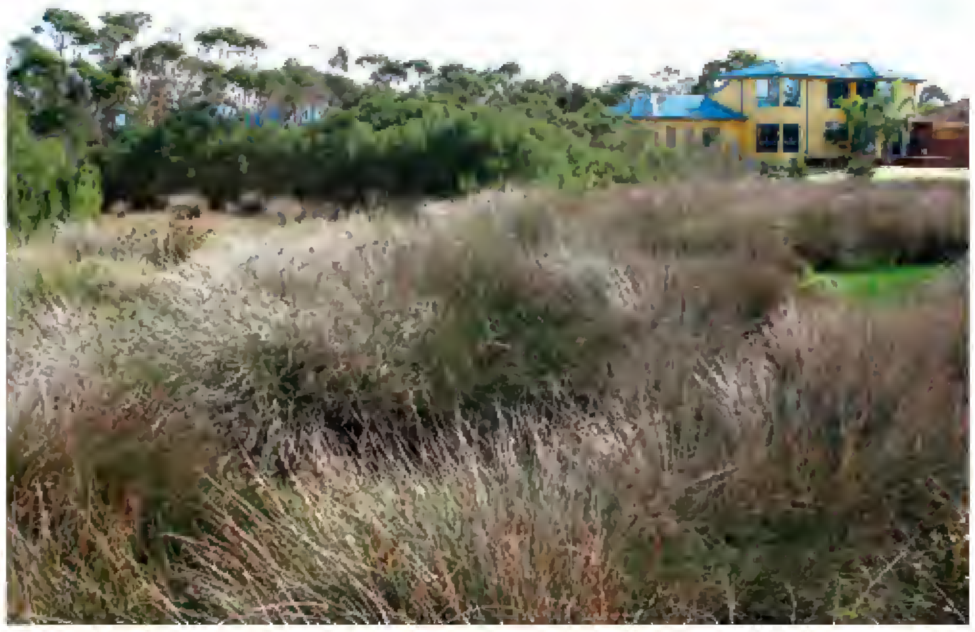

Fig. 5. Housing immediately behind coastal saltmarsh at Hastings, Western Port.

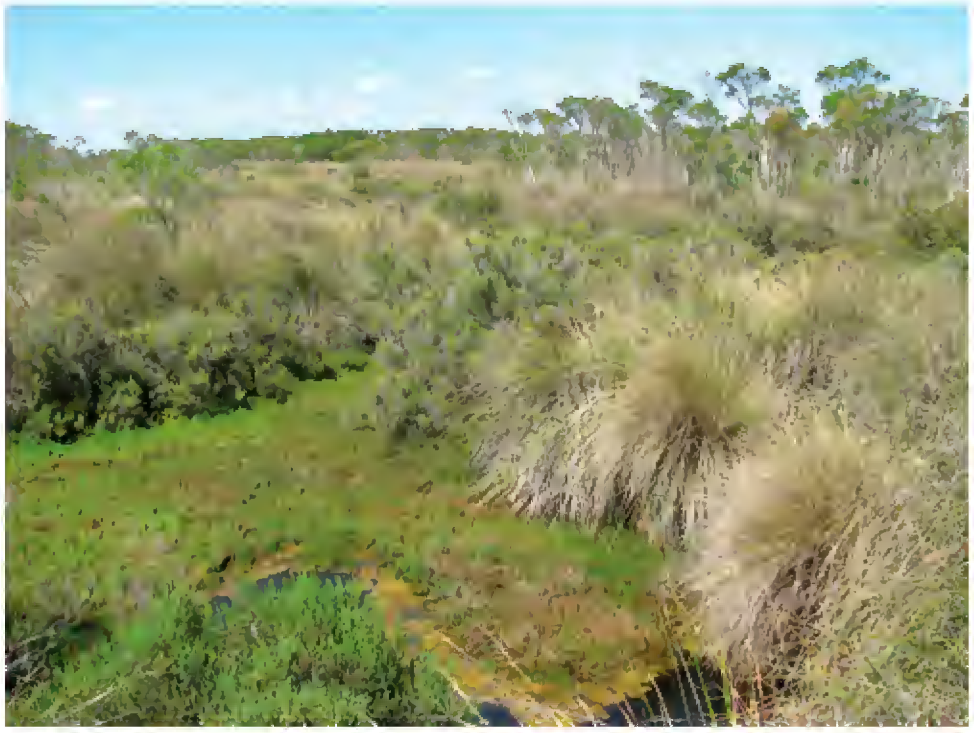

Fig. 6. Photograph showing the structural and floristic complexity of Victorian coastal saltmarsh.

Table 4 also demonstrates that historical changes to coastal wetlands have not been evenly spread along the coast: some sectors have experienced relatively little change since the mid 19th century, whereas others have been severely depleted. Proportionally large depletions have occurred, for example, in the Lonsdale Lakes area on the Bellarine Peninsula (approximately 60% depletion), the western shore of Port Phillip Bay (50% depletion), and the Powlett-Kilcunda and Anderson Inlet sectors of the South Gippsland coast (60–65% depletion). In some areas of the coast (e.g. Glenelg and parts of the Gippsland Lakes), there may have been an expansion of coastal marsh since European colonization. Figure 2 shows as examples pre-European and present-day marsh distributions for two areas of the Victorian coast: the Bellarine Peninsula and Anderson Inlet.

The following section details some of the site-specific matters that are relevant for each of the coastal sectors used in the analysis. Bioregions are shown in brackets and abbreviated as per Table 3. The discussion concerns primarily extent of marsh and does not address possible change in ecological condition; the latter could not be easily or consistently

Table 4: Estimated areas of Mangroves (M) and Other Coastal Marsh (OCM: mostly a combination of Coastal Saltmarsh and Estuarine Wetland) in various sectors of the Victorian coast before European colonization (pre-1750 area) and as at ~2008 (present-day area). All values for pre-European areas have been rounded to the nearest 10 ha and, for depletion figures, to the nearest 5%; present-day areas are known with better accuracy and are thus shown to the nearest ha. Two scenarios are given for the two coastal sectors associated with the Gippsland Lakes (Lake Wellington; and Lakes Victoria and King) as described in the text. Any recorded expansion in marsh area from pre-European times is shown simply as >100 rather than as a numerical value. ? = unknown. See text for discussion of limitations.

Sector	**Pre-1750 Area (ha)**			**Present-day area (ha)**			**Retention (%)**		
	M	**OCM**	**Total**	**M**	**OCM**	**Total**	**M**	**OCM**	**Total**
Glenelg		40	40		54	54		>100	>100
Fawthrop-Belfast		120	120		67	67		55	55
Western estuaries		590	590		606	606		100	100
Aire-Gellibrand		220	220		222	222		100	100
Surf Coast estuaries		30	30		31	31		100	100
Breamlea		410	410		363	363		90	90
Connewarre-Barwon	40	2230	2270	49	2023	2073	>100	90	90
Lonsdale Lakes		390	390		162	162		40	40
Salt Lagoon		40	40		36	36		90	90
Swan Bay		560	560		483	483		85	85
Mud Islands		40	40		36	36		90	90
Port Phillip	?	3710	3720	6	1767	1773	?	50	50
The Inlets	10	110	110	8	59	66	80	55	60
Western Port	1320	1460	2790	1230	1287	2516	95	90	90
French Island	480	1010	1500	475	1002	1476	100	100	100
Rhyll Inlet	90	130	230	93	119	212	100	90	95
Lang Lang coast		20	20	2	29	31	>100	>100	>100
Bass River	10	230	240	15	165	180	>100	70	75
Powlett-Kilcunda		200	200		75	75		35	35
Anderson Inlet	130	1120	1250	158	434	592	>100	40	45
Shallow Inlet	170	300	480	0	180	180	0	60	40
Wilsons Promontory	50	130	190	54	135	189	100	100	100
Corner Inlet	1060	1350	2410	846	513	1358	80	40	55
Nooramunga coast	1020	2820	3840	994	2263	3257	95	80	85
Nooramunga islands	1240	2100	3350	1247	2236	3483	100	>100	>100
Jack Smith Lake		1920	1920		1839	1839		95	95
Lake Reeve		3530	3530		3043	3043		85	85
Lake Wellington - Senario 1		5560	5560		3638	3638		65	65
Lake Wellington - Senario 2		480	480		3638	3638		>100	>100
Lakes Victoria and King - Senario 1		4940	4940		4019	4019		80	80
Lakes Victoria and King - Senario 2		2510	2510		4019	4019		>100	>100
East Gippsland inlets		1160	1160		995	995		85	85
Total - Senario 1	**5650**	**36480**	**42120**	**5175**	**27878**	**33053**	**92**	**75**	**80**
Total - Senario 2	**5650**	**28970**	**34620**	**5175**	**27878**	**33053**	**92**	**95**	**95**

inferred by the methods used in our study. Note also that the whole-of-State perspective inevitably leads to a glossing-over of some losses that were of comparatively minor extent; such a bias should not be seen to trivialise the smaller losses that have taken place since European colonization.

Glenelg (Brid)

This small, isolated sector has always contained a relatively small area of coastal marsh, and much of it remains intact. Field work indicated that areas of Estuarine Wetland and Estuarine Reedbed (EVC 10 and EVC 952, respectively), has apparently been converted to pasture, particularly between the Glenelg River and the South Australian border. It is possible that there has been a minor increase in the total area of coastal marsh in this sector (Table 4).

Fawthrop-Belfast (Fawthrop VVP; Belfast WaP & VVP)

Field work and several early plans indicate that significant change has occurred to the extent of intertidal marsh in this relatively small area (Anon c.1843, 1843, 1850, Barrow undated, Myers 1843). Fawthrop Lagoon near Portland has suffered very considerable hydrological change through channelling, changes to its catchment, and the conversion of the outlet to an artificial channel. The freshwater-saltwater boundary is now sharply defined by a causeway. Despite these changes, the basic shape of the lagoon probably remains the same in the saltmarsh zone (Stanley 1869*a*). There is no evidence for a substantial loss of coastal saltmarsh here, but it is likely that the brackish zone was once larger (indicating a likely loss of EVC 10 Estuarine Wetland). Since the mouth of the Moyne River has been permanently open at Port Fairy, Belfast Lough has been permanently exposed to the Southern Ocean. It now contains meadows of seagrass in the sub-tidal zone (*Zostera muelleri*, Ball & Blake 2009) and large expanses of saltmarsh. This wetland, however, was shown in 1843 as being 'fresh during the winter months' (Myers 1843). Although that evidence is cursory, it suggests that the wetland has become saltier, which probably allowed saltmarsh to increase at the expense of EVC 10 Estuarine Wetland (we have assumed half the area was once Estuarine Wetland for the purposes of area calculations.) As well, the wetland has been drained around its margins.

Western estuaries (WaP)

This sector includes the estuaries of the Surrey, Fitzroy, and Curdies Rivers, along with the intermittently open and closed coastal lagoon Lake Yambuk, in the west of the State. All are located in an agricultural landscape in which native vegetation has been almost entirely removed, including formerly vast areas of freshwater wetland. Extensive field work for this and a previous project (Sinclair & Sutter 2008) indicate that intertidal wetland vegetation was naturally restricted, and its post-European loss has been very small and proportionally relatively minor. The direct historic record, however, is sparse (Barrow 1853, Urquart undated, Anon 1854, Watson 1855) and field work suggests coastal marsh was probably restricted to small patches in Rutledge's Cutting and at the original mouth of the Merri River (near Thunder Point, Warrnambool), at the mouth of the Fitzroy River, and the lower portions of Lake Yambuk. In all cases, only *Sarcocornia quinqueflora*– and tussock-dominated (*Gahnia filum* or *Austrostipa stipoides*) saltmarsh is present, along with EVC 10 Estuarine Wetland. Most of these patches remain and have suffered only modest losses since European colonization. As shown in Table 4, we estimate the pre-European extent of non-mangrove coastal marsh as 590 ha and the current-day extent as 606 ha.

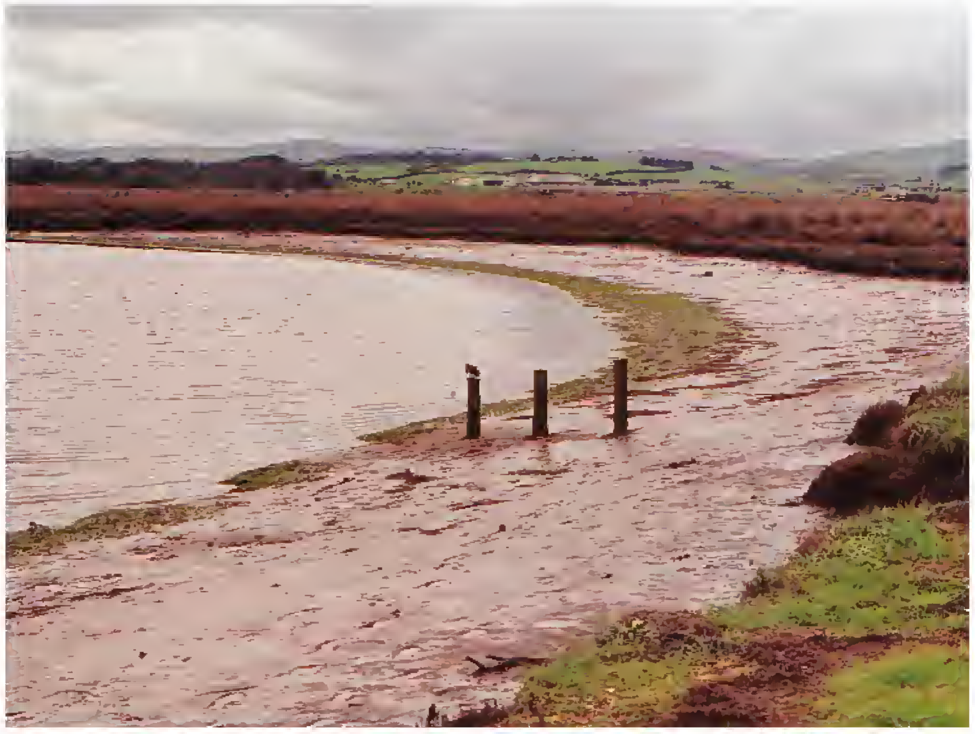

Fig. 7. Coastal saltmarsh at the mouth of the Powlett River in South Gippsland.

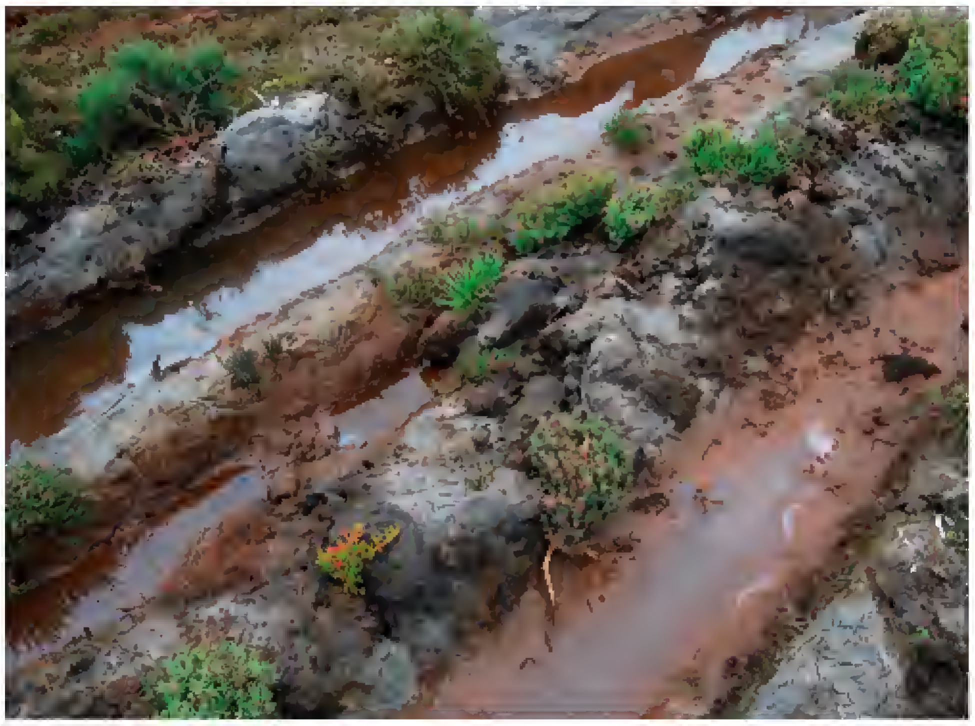

Fig. 8. Small-scale disturbance created in coastal saltmarsh by vehicular traffic. Note the new habitat created in the pools left by the vehicle tracks.

The loss of Estuarine Wetland is difficult to assess, particularly as sparse stands of *Juncus kraussii* may remain in grazed pasture after the removal of many other species that may be associated with a range of EVCs other than Estuarine Wetland (e.g. EVC 952 Estuarine Reedbed, EVC 953 Estuarine Scrub). Losses may have occurred in the hind-dune marshes on the Fitzroy River, and possibly also on the lower Merri River, although extensive drainage works between 1859 and 1870 and sand movement have changed the system radically (Anon 1854, Sinclair & Sutter 2008). All the losses in this sector of the coast are related to pasture creation, through drainage works and/or through continual grazing by stock.

Aire-Gellibrand (Gellibrand WaP; Aire OtP)

The Aire and Gellibrand River systems drain steep hills which experience some of the highest rainfall in Victoria (Table 3). The rivers discharge across large floodplains separated from the sea by natural barriers. Neither estuary currently supports mangrove vegetation, and there is no evidence that either did in pre-European times. Both river systems, however, support extensive stands of Estuarine Wetland. Field inspection and a recent mapping study of the entire Aire floodplain (Osler *et al.* 2010) confirm that extensive drainage works have radically altered its hydrological character, and led to extensive losses of native vegetation. Estuarine Wetland seems to have suffered only small declines, although it is difficult to reconstruct its exact pre-1750 distribution (Osler *et al.* 2010).

Surf-coast estuaries (OtP)

This sector of the coast supports only small patches of coastal marsh vegetation and has suffered some significant changes since European colonization, and there seems not to have been much loss of coastal marsh (Table 4). The Erskine River at Lorne retains much of its Estuarine Wetland. Painkalac Creek at Aireys Inlet also remains largely intact and supports mostly a *Sarcocornia quinqueflora*–dominated saltmarsh herbland and EVC 10 Estuarine Wetland. Changes have been greater on the Anglesea River. Here, a section of coastal marsh on the western side of the inlet was filled and converted to an urban park and roadway (Osler *et al.* 2010). Interestingly, the Anglesea River has recorded an increase in EVC 10 Estuarine Wetland, which has invaded areas of former Estuarine Scrub that were burnt and bulldozed in the 1983 fires (Osler *et al.* 2010). The tiny areas of saltmarsh and EVC 10 Estuarine Wetland on the Spring Creek at Torquay have been fragmented and reduced in area by incremental works on the shoreline, plantings and weed invasion. Overall, however, there has been little or no change in the area of coastal marsh in this sector since European colonization.

Breamlea (OtP & VVP)

Early plans (Smythe 1847, Daintree *et al.* 1863) show that the extent of coastal saltmarsh across most of this sector also has changed little since the early days of pastoral settlement, despite a history of varied and intensive land use. The eastern end of the marsh at Breamlea abuts a wastewater-treatment plant, which has destroyed a portion of the original wetland. The other substantial loss of coastal marsh occurred as a result of the landfill located at the south-western end of the sector, and a lagoon which was once located between Salt Swamp (Connewarre-Barwon sector) and Breamlea has been destroyed as well. Other incremental losses have occurred as a result of drainage for pasture, particularly in the north and centre of the sector. We estimate the total loss of coastal marsh to be of the order of 10% (Table 4).

Connewarre-Barwon (OtP & VVP although not mapped as such)

The lower Barwon River system remains a formidable and complex area of coastal marsh, despite its proximity to the large cities of Melbourne and Geelong and the long regional history of European land use. The historical record is detailed (Smythe 1847, Byerley 1855, Daintree & Ross 1862*a*) and shows that significant losses of marsh have occurred as a result of drainage for pasture in the south-western corner, at the southern boundary of Salt Swamp, along the course of the Barwon River (notably the southern side), and on the eastern side of Lake Connewarre. Comparatively minor losses have occurred elsewhere through dumping of material, development and weed invasion. Interestingly, it is possible that some small (<10 ha) gains in saltmarsh extent have also occurred since colonization, with areas on Paceys Island which were once wooded with EVC 953 Estuarine Scrub (probably *Melaleuca lanceolata*) having been converted to ‘saltmarsh’ through loss of their overstorey (Byerley 1855). The overall loss of non-mangrove marsh however, is likely to be small and, as with the previous sector, only of the order of 10%. The area of mangroves may have expanded by 20%, but as the original area was small (40 ha) this represents an increase of <10 ha (Table 4).

Lonsdale Lakes (OtP)

This section of the coast has suffered significant losses (approximately 60%) of a diverse range of saltmarsh types from the time historical data are available (Hoddle 1843, Byerley 1855, Daintree & Shepherd 1863*a*). The losses result from the accumulation of many local changes involving drainage, conversion to pasture, and the disruption of natural hydrological regimes through the construction of roadways. The Lonsdale Lakes were once more-or-less joined to Swan Bay by an expanse of marsh vegetation. Only a small section of this on the margin of Swan Bay is recorded as ‘samphire’ by Hoddle (1843) (i.e. succulent chenopods); the remainder was coastal marsh of unknown type. The loss of this area, apparently partly filled in to make way for suburban Point Lonsdale, is recorded in this sector, not Swan Bay.

Salt Lagoon (OtP)

Despite being surrounded by semi-urban development, agricultural land and roads, Salt Lagoon has largely retained its former shape and area (Daintree & Shepherd 1863*b*). All losses are of very minor extent.

Swan Bay (OtP)

Sand movement has altered the entrance to Swan Bay markedly since European colonization, partly as a result of natural processes, partly due to coastal engineering works. It is difficult to tell from the primary historical resource (Hoddle 1843, CHW Ross 1859, Cox & Stanley 1874, numerous maps of Port Phillip Bay) how much change in marsh extent these developments have caused. Although we assumed no change in extent at the entrance, saltmarsh has certainly been lost at the south-western end of Swan Bay because of the construction of roads and railways. Some small losses have occurred to the north and north-west as a result of drainage for pasture, but the saltmarsh was evidently always narrow in this region, as the hinterland rises quickly. The Edwards Point area retains essentially all its pre-European area of marsh. We estimate the overall loss in this sector to be about 15% across all marsh times.

Mud Islands (No bioregion assigned)

Due to their isolation within Port Phillip Bay, Mud Islands have been largely free from the development pressures that have led to the destruction of so many other coastal marshes in Victoria. But this is not to say this area has been static: historical records collated by Yugovic (1998) demonstrate that natural geomorphic processes have changed the shape of the islands and the marsh greatly since the early 19th century. Even so, we have assumed that there has been no net loss of saltmarsh vegetation since European colonization.

Port Phillip (VVP & OtP)

This sector has suffered massive losses of coastal marsh, brought about by intensive land-use, particularly ponds for sewage water treatment and salt production along the western shore. Fortunately the historic record is extensive and detailed (Russell 1837, Anon 1841, Garrard & Shaw 1850, Christie 1853, Ross 1859, Daintree & Ross 1862*b*, Cox 1864, Noone 1880*a*, 1880*b*, Goldsmith & McGauran 1896, Proeschel undated). The marshes which originally occurred in inner Melbourne and the port area have been destroyed by the growing city. The marshes of Altona retain much of their area, but have been corralled within a largely suburban landscape. The Cheetham wetland at the mouth of Skeleton Creek has been almost entirely converted to evaporation ponds, and the upper marsh has been replaced by the Sanctuary Lakes housing development. The once extensive string of marshes that stretched between the Werribee River and Point Wilson has been obliterated by the Western Treatment Plant. Similarly, the large marshes between Point Wilson and Limeburners Bay, along with the marshes of Point Henry, have been almost fully converted to salt ponds. Only the RAAF Lake and The Spit (Point Wilson) remain as large, relatively intact marshes in this part of the Victorian coast. Overall, we estimate the loss of coastal marsh (mainly saltmarsh rather than mangroves) at approximately 50%.

The Inlets (GipP)

The Inlets have lost up to 40% of their original coastal marsh, mainly as a result of the loss of saltmarsh and estuarine wetlands rather than of mangrove (Table 4).

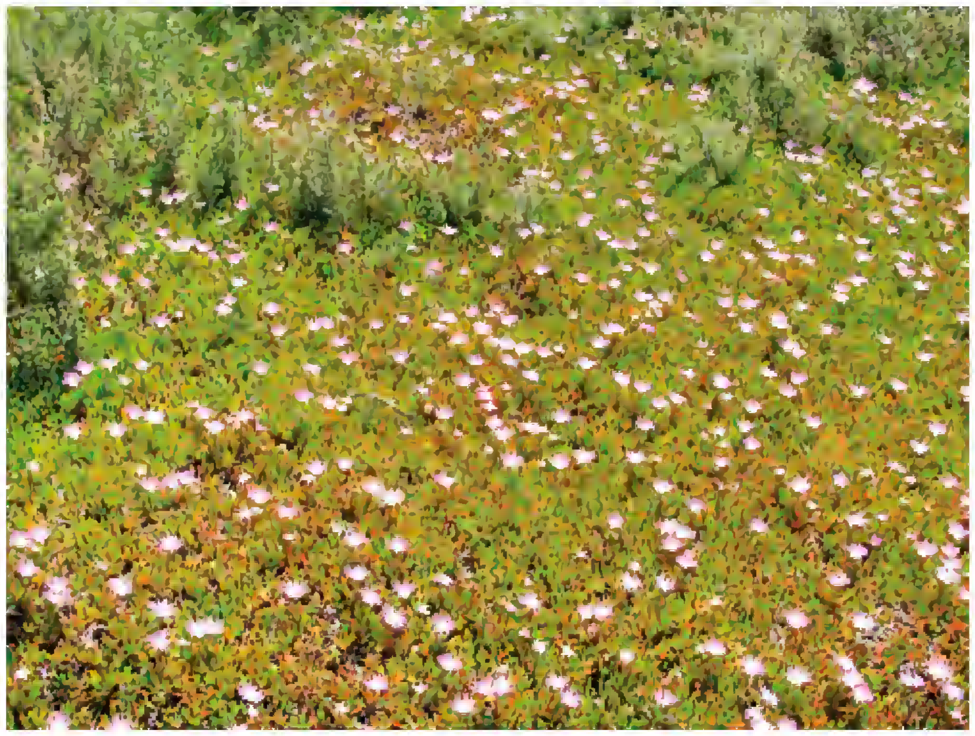

Fig. 9. Flowering Rounded Noonflower, *Disphyma clavellatum.*

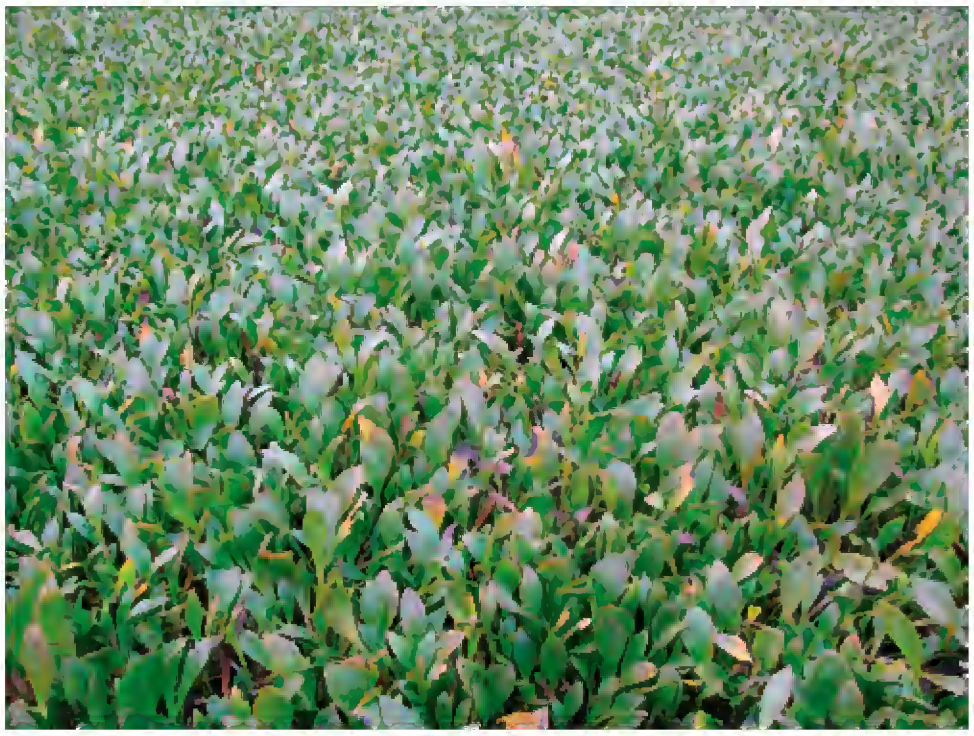

Fig. 10. Extensive bed of Swampweed, *Selliera radicans.*

Today they present an unusual superimposition of the past and present. The once vast Koo Wee Rup swamp now discharges via large drains into Western Port. However, between the Inlets the imprint of the original tidal channels and portions of coastal marsh and other vegetation remains intact. The marsh is diverse, containing EVC 10 Estuarine Wetland, EVC 140 Mangrove Shrubland and EVC 9 Coastal Saltmarsh Aggregate of many kinds, including *Sarcocornia quinqueflora* herbland, *Tecticornia arbuscula* shrubland, and tussock-dominated saltmarsh of *Gahnia filum* or *Austrostipa stipoides*. The largest losses have been west of Lyall Inlet, whereas most of the marsh east of Lyall inlet apparently remains intact (Smythe 1842*a*, Yugovic & Mitchell 2006). This is not to say, however, that vast areas of other types of wetland (e.g. EVC 53 Swamp Scrub) have not been lost following the drainage of Koo Wee Rup Swamp (see Introduction).

Western Port coast (GipP)

The Western Port coast retains a massive area of intact coastal marsh, including EVC 140 Mangrove Shrubland, EVC 10 Estuarine Wetland and EVC 9 Coastal Saltmarsh Aggregate dominated either by *Sarcocornia quinqueflora* or *Tecticornia arbuscula*. It has also suffered local historical losses as a result of infilling and drainage, and agricultural, urban and industrial development. The primary historic resource is good for Western Port, and consists of several early survey plans and aerial photography taken prior to extensive land-building (Smythe 1842*a*, 1842*b*, 1842*c*, 1842*d*; Bailliere 1866, Air Photo Westernport (868) A2 Central Plans Office neg. No. 20898 (1958)). The construction of the Hastings foreshore and marina resulted in the destruction of approximately 33 ha of saltmarsh, and the industrial developments at nearby Long Point resulted in the destruction of a further 23 ha. HMAS Cerberus near Sandy Point also required minor land-claims into former saltmarsh. Apart from these major developments, infilling and drainage for pasture have also destroyed large expanses of marsh, particularly from Watson's Inlet around to Yallock Creek; and near Stockyard Point. In contrast, the marshes on Phillip Island and the Gurdies–Grantville coast remain largely intact despite being bordered by private land, and have suffered only slight losses through drainage and grazing. This retention of coastal wetland in the bioregion is presumably a function of the slightly steeper terrain at the inland border of the marshes. We estimate that overall losses have been proportionally small, probably <10%. Even so, because of the large original area of coastal marsh the total area thought to have been lost is substantial, at over 250 ha (Table 4).

French Island (GipP)

The vast stands of pre-European coastal marsh and mangroves on French Island remain almost entirely intact (Smythe 1842*b*; Bailliere 1866). Some losses have occurred as a result of clearing of Mangrove Shrubland (e.g. see Lacey 2008), and some estuarine wetland has been converted to pasture in the upper Redbill Creek; extensive artificial ponds have modified but not removed much of the saltmarsh on the north-eastern coast.

Rhyll Inlet (GipP)

Although it retains substantial areas of coastal marsh (mostly Mangrove Shrubland and *Tecticornia arbuscula* Coastal Saltmarsh), Rhyll Inlet has suffered losses since European colonization. The rubbish tip on Cowes-Rhyll Road has obliterated 5 ha of coastal marsh, and extensive drainage works on the eastern end near Cowes have destroyed more. A detailed early map (Smythe 1842*b*) suggests that the spit sheltering Rhyll Inlet has lengthened since colonization, and there may have been some modest expansion of coastal marsh at the eastern end of the inlet. Overall, losses have probably been <10% of the pre-European area (Table 4).

Lang Lang Coast (GipP)

The Lang Lang coast is unusual in that it shows a likely increase in the extent of coastal marsh since European colonization (see also Lake Wellington and Lakes Victoria and King). At settlement, this section of the Victorian coast contained virtually no saltmarsh or mangroves, and was fringed instead with a dense stand of Swamp Paperbark *Melaleuca ericifolia* where the Tobin Yallock Swamp directly met the sea (Smythe 1842*c*; Yugovic & Mitchell 2006). Fresh water spilled from this swamp, an event described by Smythe (1842*c*) as 'numerous rills of freshwater continually running'. The coast at this time was probably cliffed and eroding (Yugovic & Mitchell 2006). Drainage, however, has left the former landscape unrecognisable. A series of bund walls now line the coast, and fresh water is channelled to the sea. A band of saltmarsh has formed on and in front of the bund walls, above the intertidal zone, and presumably receives salty water from ocean spray. This marsh is mostly species-poor *Sarcocornia quinqueflora* herbland. Small areas of remnant marsh occur only at the abandoned mouth of the old Yallock Creek.

Bass River (GipP)

Pasture-creation has destroyed a large proportion of the marsh on the Bass River estuary, with almost the entire margin being bounded by walls, drains or fill. The invasive Cord-grass (*Spartina* spp.) grows in extensive mats in channels and on mudflats throughout the marsh, and has resulted in the exclusion of native-dominated marsh in places. Interestingly, an early plan of Western Port (Smythe 1842*c*) does not show any mangroves at the mouth of the Bass River, despite showing them elsewhere, including nearby at Settlement Point. Whether the extensive stands which now exist are a recent invasion is doubtful, and it is conceivable that the survey plan was incomplete. Overall, losses of coastal marsh may have been of the order of 25% (Table 4).

Powlett-Kilcunda (GipP)

The small portion of marsh at Kilcunda, supporting a range of saltmarsh types and EVC 10 Estuarine Wetland, retains most of its former extent. The Powlett River also retains most of its original saltmarsh, but large areas of Estuarine Wetland have probably been lost to pasture on the large plain to the north east of the river mouth. Proportionally, this sector of the coast has suffered among the greatest losses of coastal wetland; we estimate the loss to be about 65%, or about 125 ha (Table 4).

Anderson Inlet (GipP)

Anderson Inlet has experienced some of the largest absolute and proportional losses of coastal marsh anywhere in Victoria since surveys were first undertaken in the 19th century (Smythe c.1847; Stanley 1869*a*, 1869*b*, Anon undated *a*). Much of the saltmarsh and mangrove areas have been destroyed following extensive drainage works for conversion to pasture, and we estimate that only 40% of the original non-mangrove marsh remains (Table 4). Losses have been very substantial at the eastern end of the Inlet, and around the mouth of the Tarwin River. Moving sands also have had a minor impact on saltmarsh extent near Point Smythe. The spread of *Spartina* in the Inlet ensures that changes continue (Blood 1996).

Shallow Inlet (GipP)

This section of the coast has also undergone substantive changes since European colonization, representing some of the largest losses of coastal marsh in Victoria. Land-building works have destroyed vast areas of marsh, particularly on the southern side of the inlet. Bird (1993, Figure 167) shows the loss of wetland that has occurred since European colonization. Interestingly, both the early plans consulted for this part of the coast show extensive areas of both 'mangroves' and 'samphire' (i.e. saltmarsh) (Smythe 1848, Anon undated *a*). There are currently no mangroves in Shallow Inlet, and unless we assume both early plans were in error, it would seem that the prior stands (perhaps ~170 ha in total) have been entirely lost.

Wilsons Promontory coast (WPro)

This small sector is entirely within a National Park and is bordered by intact native vegetation. The Park was reserved in 1898, gazetted in 1905, and progressively added to in 1908, 1928, 1947 and 1969. Although there has been sporadic use of the Park's terrestrial and coastal resources (Parks Victoria 2011), it seems to be the case that coastal saltmarsh or mangroves have suffered no measurable decline in extent.

Corner Inlet coast (GipP)

Some of Victoria's largest losses of saltmarsh have occurred in this sector, and we estimate that only about 45% of the pre-European area of coastal marsh remains (Table 4). Losses have been mainly of saltmarsh rather than of mangrove, and about 80% of the pre-European area of the latter remain today. Most of the existing saltmarsh is bordered by channels and walls/levees, and the upper marsh has been drained and converted to pasture. Unfortunately the historic record seems to lack detail at a fine scale (Smythe 1848, undated *a*, Wright 1849, Anon undated *b*), however field observations, aerial photographs and elevation data come together to provide strong and consistent evidence of former marsh extent. For the purpose of area calculation, we delineated a putative pre-

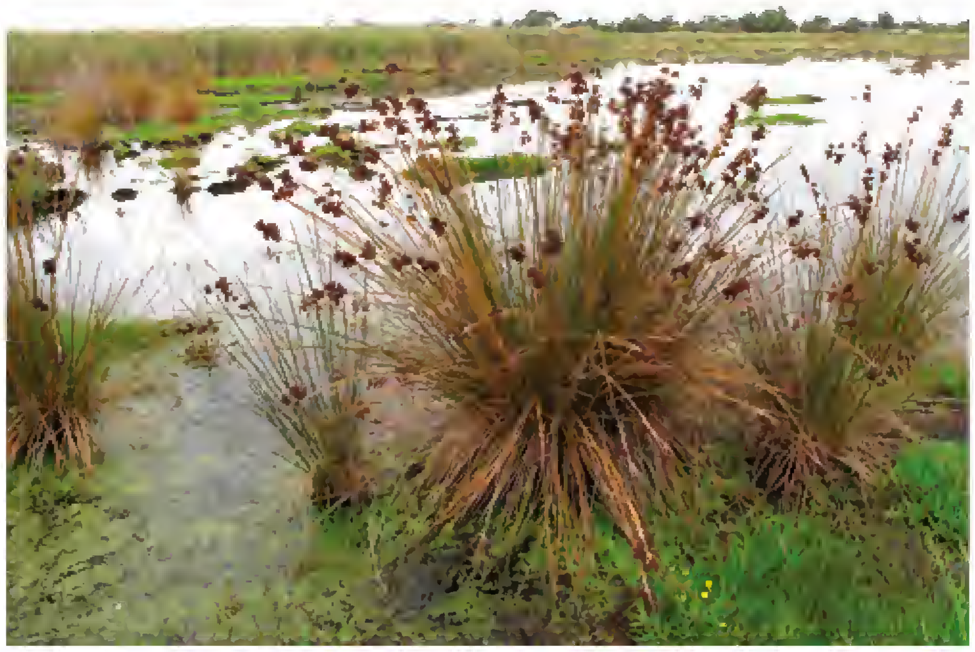

Fig. 11. *Juncus acutus*, an exotic weed now found in many coastal or salinized wetlands in Victoria. This photograph is of wetlands in the Seaford-Frankston Swamp area to the south of Melbourne.

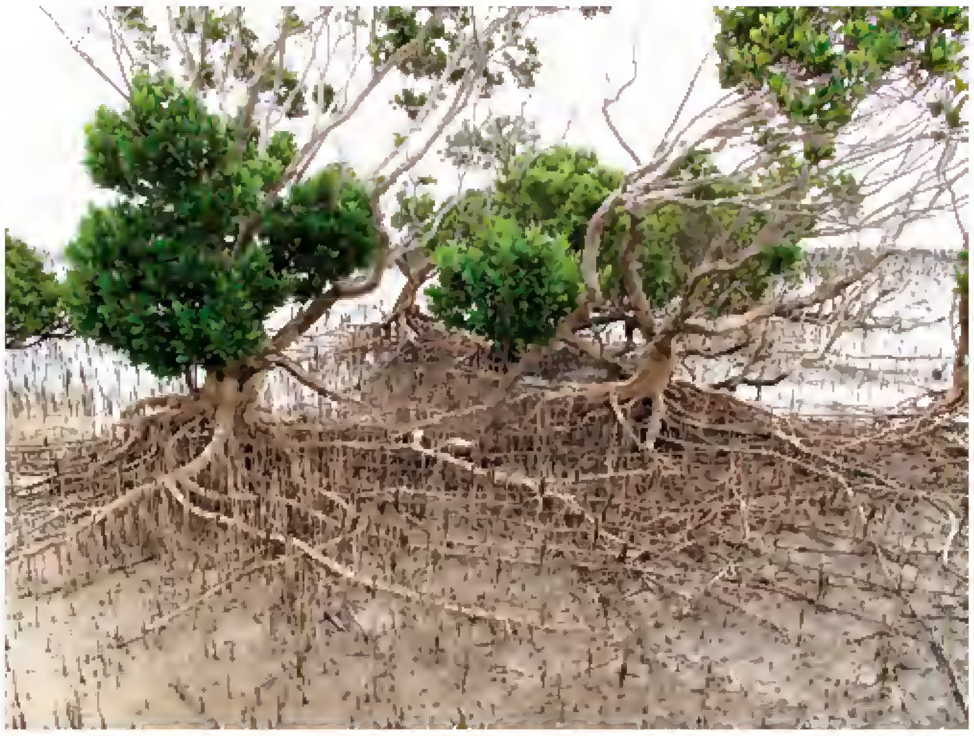

Fig. 12. Roots and pneumatophore of the mangrove *Avicennia marina* exposed by coastal erosion in the Grantville area of Western Port.

1750 boundary between mangroves and saltmarsh, using intact areas to guide us in the relative widths of these bands.

Nooramunga coast (GipP)

Although the historic record is detailed around Port Albert (Anon 1842, Smythe 1843, Anon undated *c*), it is poorly resolved elsewhere in the sector. Like Corner Inlet, however, much of the marsh is now bounded inland by sea walls and channels, which provide direct evidence of the extent to which the landscape has changed. Losses of mangroves have probably been of the order of 5% and of other coastal marsh types of up to 20% (Table 4).

Nooramunga islands (GipP)

The Nooramunga islands remain almost entirely uncleared and undeveloped and, as such, probably represent the least disturbed large sector of coastal marsh on the Victorian coast. The marshes include extensive stands of EVC 140 Mangrove Shrubland, and EVC 9 Coastal Saltmarsh Aggregate mostly dominated by *Tecticornia arbuscula*, *Sarcocornia quinqueflora Gahnia filum* or *Austrostipa stipoides*. Although we are aware that geomorphic changes have taken place in the area since settlement, the historic record does not allow an accurate reconstruction to be made of past patterns (Smythe 1848, Wright 1849, Smythe undated *b*, Anon undated *b*); thus we have assumed that the extent of marshes in this area has remain unchanged since European colonization.

Jack Smith Lake (GipP)

The historic record for Jack Smith Lake is poor (Smythe undated *d*), and field inspection and aerial photograph interpretation suggest that much (95%) of the marsh area remains, but with small areas lost to infilling, drainage and grazing.

Lake Reeve (GipP)

Lake Reeve itself retains most (~85%: Table 4) of its former area of marsh, with the exception of one large area claimed for pasture at the western end. The historic record is detailed and shows the vegetation has otherwise changed very little in its extent since the early 19th century (Smythe undated *c*, undated *d*, undated *e*, Anon undated *d*). Perhaps the biggest losses have been outside Lake Reeve itself, but included in this sector: Smythe (undated *c*) shows extensive patches of 'saltmarsh' between Lake Reeve and Lake Wellington (>750 ha in total; distinguishable from areas labelled 'swamp' and 'tea-tree'). Such areas were clearly not intertidal, but presumably interacted with saline groundwater. Similar patches of 'coastal' marsh lacking tidal connection exist elsewhere (e.g. Lonsdale Lakes). The examples near Lake Reeve have largely been converted to pasture and, in one case, a farm dam.

Lakes Victoria and King (GipP)

The ever-shifting natural entrance to the Gippsland Lakes from Bass Strait was replaced by a permanent opening as a result of engineering works between 1870 and 1889 (Bird 1965, Bird & Lennon 1989). Bird (1966) marshalled evidence to indicate that the creation of the permanent opening to the Southern Ocean had caused an increase in salinity in the Gippsland Lakes system, leading to an increase in saltmarsh at the expense of freshwater wetlands. His conclusions are broadly supported by our fieldwork, which showed strong evidence of dieback of the fringing freshwater and brackish-water wetland systems. Nonetheless, the historic record shows clearly that saltmarsh was present and extensive around Lakes Victoria and King in the early 19th century. Saltmarsh or salt lakes apparently occurred naturally right around the lakes: maps show them in Victoria Lagoon and the surrounding depressions, in Jones Bay (Wilkinson 1849), around Loch Sport, Beacon Swamp, Blond Bay, Point Wilson, near Paynesville, on the Mitchell Silt Jetties (Smythe undated c, undated d; Wilkinson 1849), on Raymond Island (Anon undated *e*) and on the Boole Poole Peninsula.

As noted in the Methods, two scenarios are shown for the change in coastal marsh area in Lakes Victoria and King. Under Scenario 1, we estimate that there were 4,940 ha of non-mangrove marsh prior to European colonization. About 80% remains today (Table 4). Under Scenario 2, however, the pre-European estimate falls to 2,510 ha and thus there appears to have been a very substantial increase in the area of saltmarsh (of the order of 1,500 ha) in this part of the Gippsland Lakes. It would thus appear that saltmarsh has expanded in some places around the Gippsland Lakes, remained static in others, and decreased in others. Direct losses of coastal saltmarsh, caused by local land-claims, have in comparison been minor. Given the uncertainty as to whether some large expanses of marsh are remnant or adventive, we have presented two extreme scenarios for this sector (as described above).

Lake Wellington (GipP)

Like Lakes Victoria and King, Lake Wellington and its fringing wetlands have been strongly influenced by the opening of the artificial entrance to the ocean at Lakes Entrance (Boon *et al.* 2008, Raulings *et al.* 2010). Bird (1965, 1966) suggested that Lake Wellington was once essentially a freshwater lake, and that the saltmarsh on its margins has invaded recently, replacing prior freshwater systems. The historical record partially supports this view. Myers (c.1840) shows Lake Wellington as 'fresh' (Lake Reeve is shown 'Salt'; Lakes Victoria and King are unannotated). In contrast, other early sources (Smythe undated c; Anon undated e) show very extensive areas of saltmarsh interlaced with 'tea tree' and 'swamp' on the southern margins of Lake Wellington (and Lake Coleman), and in many depressions inland between Lake Wellington and Lake Reeve (see Lake

Reeve sector, above). In his more-detailed 1965 study used as a basis for the 1966 synthesis, Bird acknowledged the natural (more inland) occurrence of these saltmarsh areas. There is, however, no evidence for an extensive pre-European occurrence of saltmarsh on the western and northern shores of Lake Wellington, which suggests that it has expanded its overall area on this part of the coast.

As for Lakes Victoria and King, we provide two scenarios for Lake Wellington in order to account for several large periods of uncertain history. There is a substantial difference in the areas of pre-European coastal marsh yielded by the two approaches: Scenario 1 gives an estimate of 5,560 ha, whereas Scenario 2 gives a much lower value of 480 ha. Which of the two scenarios is employed has a major impact on the amount of coastal marsh estimated to be lost (or gained) since the middle of the 19th century (see Table 4).

East Gippsland inlets (EGL)

Although little direct evidence of saltmarsh loss could be found for this sector of the coast in the historical record, an inspection of aerial photographs reveals land-claims on the lower Snowy River, which have apparently lead to large, local losses of coastal marsh (probably dominated by *Sarcocornia quinqueflora*) and Estuarine Wetland (Lees 1855, Anon undated *d*, undated *f*). We estimate the loss in this sector to be about 15%.

Discussion

What Australian landscapes looked like, what vegetation was present, and how it was patterned before European colonization are topics that have become of increasing interest to many researchers and natural-resource managers over recent decades (e.g. see Benson & Redpath 1997, Lunt 1997, 1998, 2002, Griffiths 2002, Hateley 2010). Concurrent with this widespread interest, maps that accurately represent vegetation patterns before European colonization also have become increasingly important to researchers and managers, and are commonly called 'pre-1750 vegetation' maps or similar. As well as being historically interesting in their own right, such maps provide a reference point to calculate relative degrees of change and depletion and are sometimes used as aspirational targets for restoration of degraded environments (Oliver *et al.* 2002). In Victoria the degree of depletion is also essential information for determining the conservation status of vegetation (the BCS, or 'Bioregional Conservation Significance') and in designing a reserve system that is 'comprehensive, adequate and representative' (Commonwealth of Australia 1997).

A tremendous conceptual difficulty with historical reconstructions of the pre-European Australian landscape is that there are 'radically different readings of the same landscape' (Horton 2000, page 72). As examples, controversy rages as to the nature of Aboriginal fires regimes and their ecological impacts (e.g. see Benson & Redpath 1997,McLoughlin 2004), the density of trees in terrestrial woodlands (Benson & Redpath 1997, Lunt 1997, 1998, Horton 2000, Griffiths 2002) and the intensity of the regrowth of so-called woody weeds (Benson & Redpath 1997). Against this background of the suite of methods available for analysis – all with limitations – is the possibility of a strongly political or ideological element to many of the interpretations that are made (e.g. see Horton 2000).

Loss of coastal marshes in Victoria

Difficulties in interpretation and their solution. In the case of coastal marshes, difficulties in interpretation are

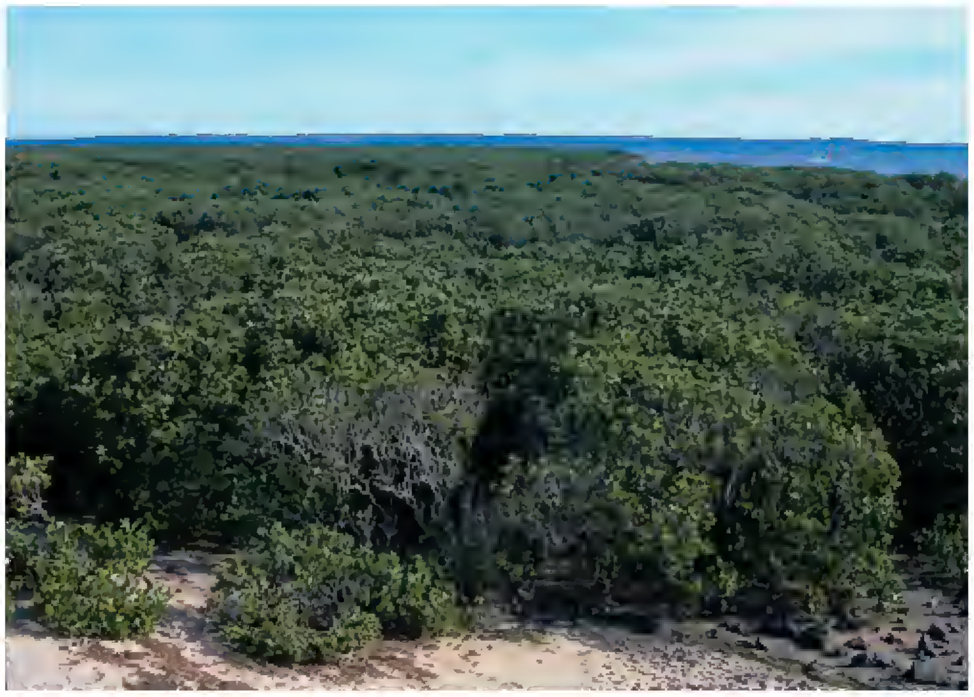

Fig. 13. Dense stands of the mangrove *Avicennia marina* along the northern shore of Western Port, near the township of Tooradin.

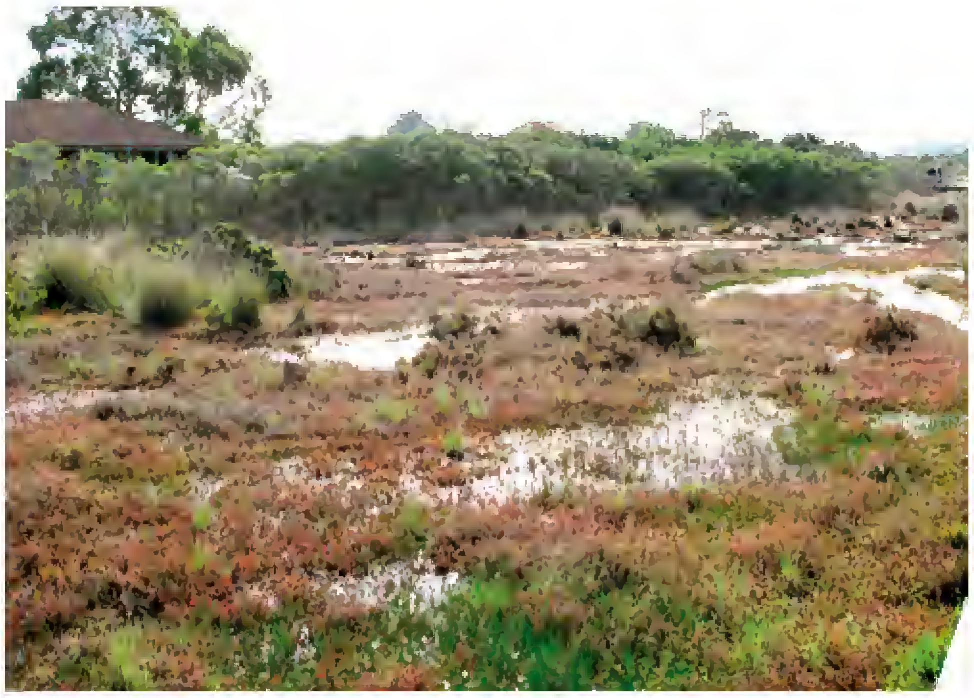

Fig. 14. Housing immediately behind coastal saltmarsh and ephemeral saline pools at Hastings, Western Port. Compare with Figure 5.

compounded by the temporal dynamism of the vegetation and its structural simplicity. In some cases vegetation change is a simple case of subtraction: vegetation has been removed *in toto* from the landscape, so that current vegetation maps are simply 'cut outs' from the pre-1750 maps. Coastal marshes probably differ significantly from that simple scenario in that they may have expanded or contracted in different parts of Victoria since colonization; in some places saltmarsh now occurs in places where it did not before colonization. Many patches of coastal marsh are also small and intricately shaped, meaning they are not easily mapped when they have been destroyed and their size can be difficult to estimate without substantial error. Furthermore, saltmarshes do not leave behind easy clues for historical ecologists to unravel. When they have been 'reclaimed', the loss means that they have often been utterly destroyed through changes to the whole landscape, such as from land-claims using earthworks, extensive landscaping for saltworks or water treatment, or the removal of whole areas of marsh for marinas, ports, oil refineries and other types of coastal development. In contrast, many other vegetation types are removed without such wholesale alteration of the landscape and small pockets or remnant often remain as a clue to the original vegetation. When forests or woodlands have been removed, for example, the soil usually remains accessible for interpretation, as do a few relict trees. In contrast, coastal saltmarsh can disappear without a trace.

Calculating the area of coastal marshland that existed before European colonization and thus the degree of retention (and depletion) is further complicated by our need to deal with an always imperfect knowledge of the past and, especially, a limited or ambiguous understanding of the vegetation patterning that is reported in the historical sources. We are particularly uncertain as to whether some areas of coastal marsh in the Gippsland Lakes region are recent incursions of coastal marsh or remnants of pre-colonial wetlands. To address this uncertainty we offer two scenarios: Scenario 1, where all the ambiguous marsh is considered to be adventive (which offsets much of the other losses across the State); and Scenario 2, where it is all considered relict. These two scenarios differ greatly and the area of uncertain origin is very large, as shown in Table 4 and discussed earlier.

Similarly, it is also not clear how best to treat small areas of land or water that occur within the current area of coastal marsh, but which are not marsh vegetation – areas such as brine pools (including EVC 842 Saline Aquatic Meadow), bare mud, small 'islands' of sand or of higher land, and man-made structures such as in-fill and levees. As these areas are clearly not to be included as 'marsh' when compiling the figures showing current costal marsh extent, they were excluded in the Victorian Saltmarsh Study (2011). Despite this, the resolution of the historic maps and plans means that such areas are mostly included within areas shown as 'marsh' on the old maps. If the current and historic figures were taken at face value, many areas of non-marsh would be excluded from the current figures but included in the historic data, which makes for an unbalanced comparison. Unfortunately, we cannot know where the areas of brine pools, sand deposits and bare mud were in the past, since these areas change over decadal time-spans. Therefore we included all areas of non-marsh in the current total for Other Coastal Marsh. This makes the figures higher than those presented originally in the Victorian Saltmarsh Study (2011), as discussed next.

Magnitude of historic losses. Notwithstanding the difficulties of interpretation, our analysis shows that prior to European colonization Victoria supported approximately 56.5 km^2 of mangrove swamps and 290–364 km^2 of Other Coastal Marsh. These values sum to a total area of 346–421 km^2 for mangrove and non-mangrove coastal marsh combined. Present-day areas for mangroves were estimated at 51.75 km^2 and for other types of coastal marsh at 278.78 km^2 (Table 4). Thus, approximately 92% of mangroves and 75–95% of Other Coastal Marsh remains today on a whole-of-State basis. When these values are conflated, we estimate that overall, 80–95% of pre-European coastal marsh remains in Victoria as of c. 2008.

The detailed mapping undertaken earlier by the Victorian Saltmarsh Study (2011) reported that there were 51.8 km^2 of EVC 140 Mangrove Shrubland in Victoria, as well as 192.1 km^2 of EVC 9 Coastal Saltmarsh Aggregate, 32.3 km^2 of EVC 10 Estuarine Wetland, and 6.5 km^2 of EVC 196 Seasonally Inundated Sub-saline Herbland. Together, the area of these non-mangrove coastal marshes sums to ~231 km^2. Not unexpectedly, there is agreement between our and the previous (Victorian Saltmarsh Study 2011) estimates of the current-day area of mangroves. The apparent discrepancy for non-mangrove coastal marshes arises because in order to allow a valid comparison to be made of current-day and pre-European data sources we had to include all areas of non-marsh in the total for Other Coastal Marsh, as described above. (The total area is calculated at 278.78 km^2 in Table 4 and is thus about 20% higher than in the earlier State-wide estimate.)

Geographic and floristic variations. Although the State-wide loss of coastal marsh has been of the order of only 5–20%, some parts of the coast have experienced relatively little change since the mid 19th century, whereas others have been severely depleted and in a few areas there has been an expansion of coastal marsh, both of mangroves and of coastal saltmarsh/estuarine wetland. As noted earlier, the situation with the Gippsland Lakes is complex and problematic, and according to the method used to interpret the original data sources, there has either been a substantial increase or a slight loss (~10–20%) in the area of coastal wetlands in this part of Victoria. In many of the coastal sectors, however, losses have been minor (<10%) and in many cases there is almost the same area of coastal marsh now as in pre-European times.

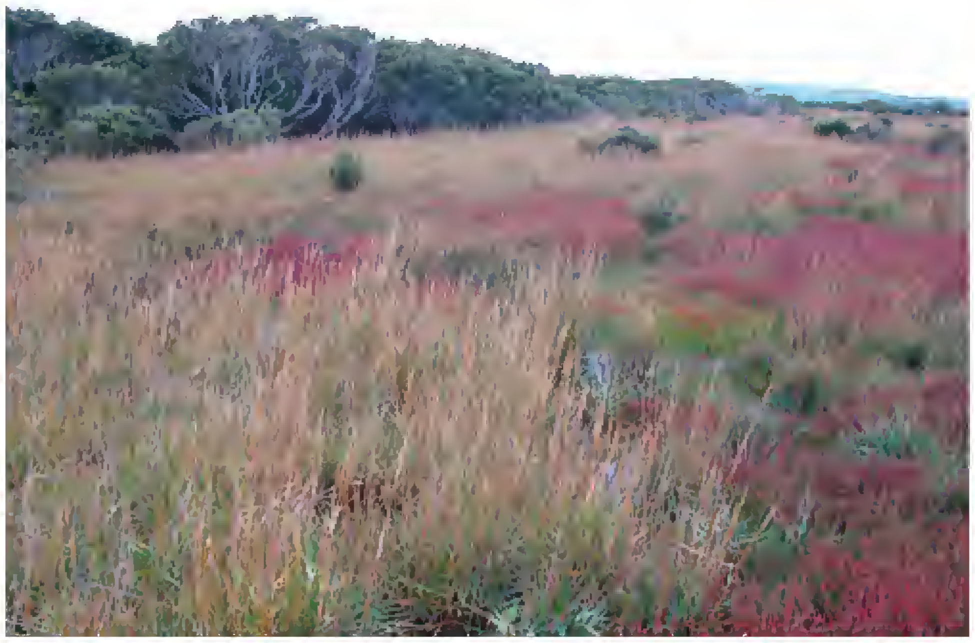

Fig. 15. The exotic Cord-grass *Spartina* growing at Anderson Inlet.

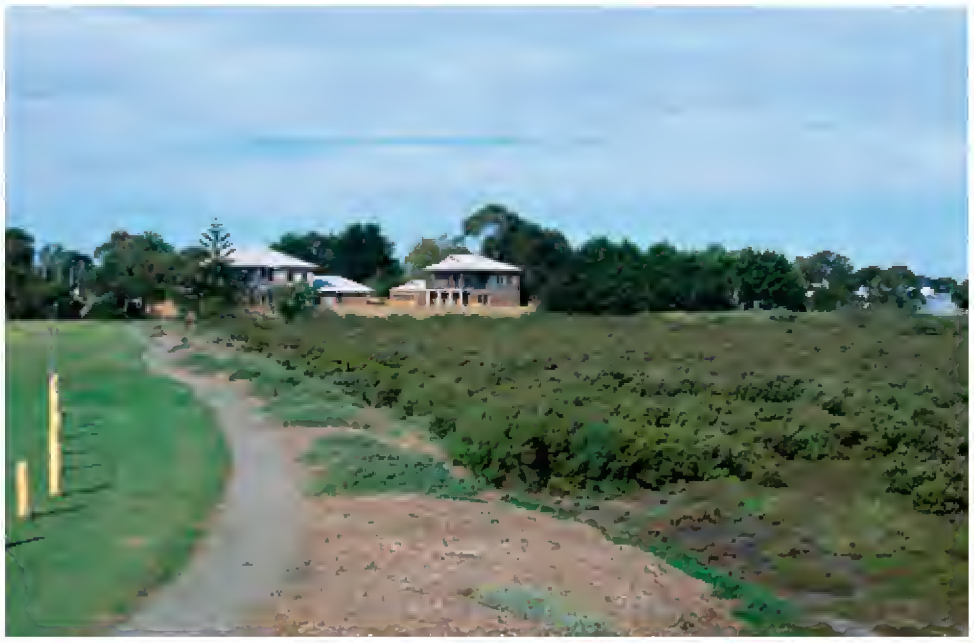

Fig. 16. Loss of coastal saltmarsh from the Hastings area of Western Port as a result of urban housing and the creation of sports fields.

A distinction has often been drawn between two general forms of saltmarsh on the Victorian coast (Barson & Calder 1981). The first, a 'dry' type, is present mostly in the central-western parts of the State, where low summer rainfall and high temperatures lead to intensely hypersaline conditions in the more elevated sites. Here the vegetation is dominated by *Tecticornia pergranulata* and *Tecticornia halocnemoides*. The second, 'wet' type, is found where rainfall is higher, mostly in the eastern part of the State, including Western Port; the vegetation in this saltmarsh type is dominated by samphires such as *Sarcocornia* spp. and *Tecticornia arbuscula*. Given the composition and ecology of coastal saltmarsh differs greatly between different sectors, our figures could mask potentially large changes for some species or saltmarsh types. Although the historic data are not sufficiently detailed to permit a quantitative state-wide analysis at the level of species or saltmarsh types, a qualitative consideration of likely losses in each sector may reveal some of this pattern.

Across the State, the largest absolute losses have probably been of EVC 140 Mangrove Shrubland, and coastal saltmarsh dominated by *Tecticornia arbuscula*, and the main losses have occurred in Lonsdale Lakes, western shore of Port Phillip Bay, Anderson Inlet, Shallow Inlet, Powlett-Kilcunda, Corner Inlet and Nooramunga. All these regions except the first two are located along the coast of Gippsland, an area of Victoria that experienced very considerable development for agriculture, mining and industry (Wells 1986, Watson 1997). The large proportional losses in western Port Phillip Bay have occurred with marshes that occur along lower-rainfall coast; here the dominant taxa are *Tecticornia pergranulata*, *Tecticornia halocnemoides* and *Disphyma crassifolium*.

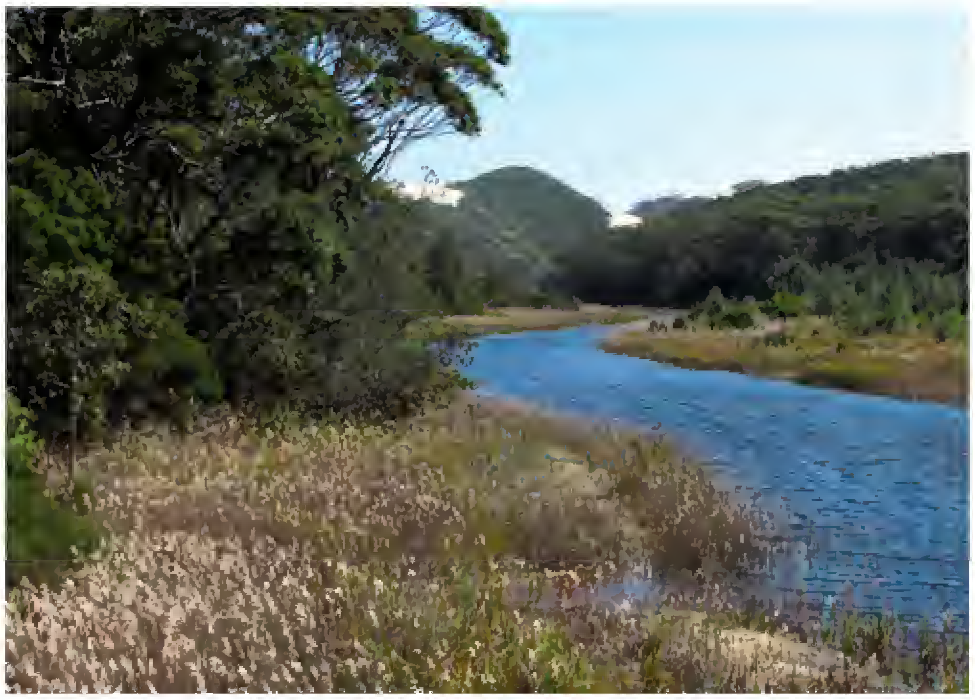

Fig. 17. One of the more pristine coastal marshlands in Victoria: the mouth of the Thurra River in East Gippsland.

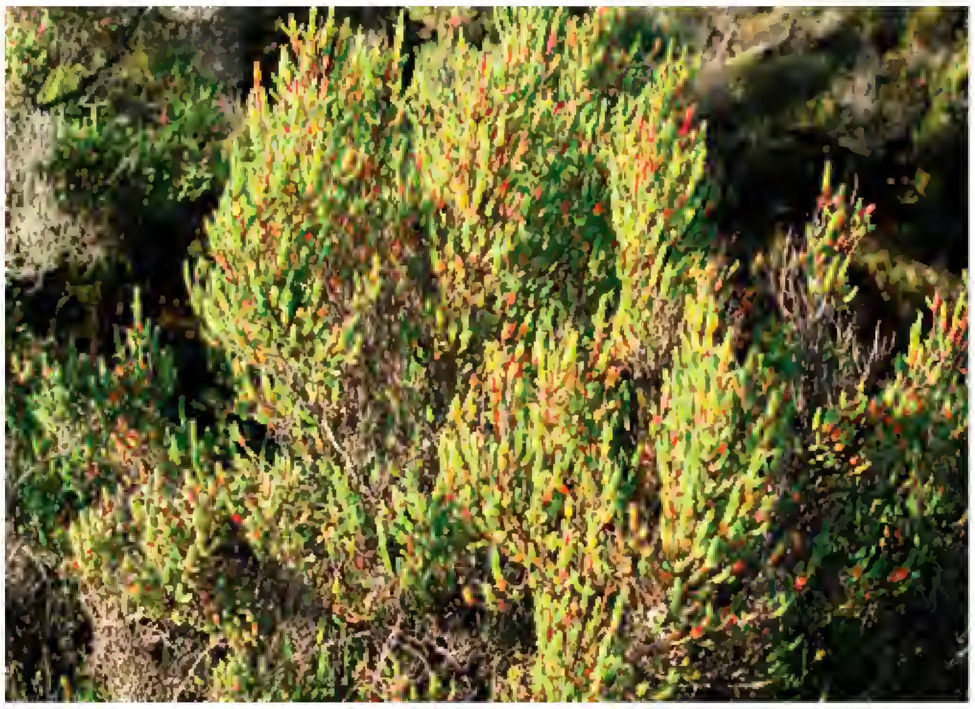

Fig. 18. *Tecticornia arbuscula* growing in a saltmarsh along the eastern shore of Port Phillip Bay.

Comparison with losses of coastal marshes elsewhere

The proportion of overall wetland loss for Victoria seem not to be as severe as have been reported for New South Wales and Queensland, where losses of 30–70% since the 1940s–1950s are not uncommon (Saintilan & Williams 2000, Harty & Cheng 2003). The comparison is, however, made difficult by the different time frames of the various investigations; our study quantified losses since the middle of the 19th century, whereas Saintilan & Williams (2000) and Harty & Cheng (2003) studied losses only since approximately World War 2, when aerial photography became widely used for landscape analysis and mapping (Fensham & Fairfax 2002).

The losses also appear to be relatively small in comparison with many Northern Hemisphere coastal systems, where losses of 20–60+% are not uncommon (Doody 2008; Gedan & Silliman 2009). In addition to the statistics cited in the Introduction, it may be noted that, in the USA, ~50% of the original saltmarsh of Narragansett Bay has been filled in, 73% of the original saltmarsh lost from Puget Sound, and 95% from San Francisco Bay (Doody 2008). Using historical maps, Bromberg & Bertness (2005) concluded that the average loss of New England (USA) marshes since the early 1800s was 37%; Rhode Island had lost 53% of its coastal saltmarshes since 1832, and the Boston area had lost 81% since the late 1700s.

The apparently small overall proportional loss in Victoria compared with other coastal areas is attributable to a number of factors. The first is that a great deal of coastal marsh occurs on relatively inaccessible islands, notably in the Corner Inlet–Nooramunga area. The second is that Victoria has had a history of agricultural development and population growth that is much shorter than those of coastal areas in the Northern Hemisphere. Furthermore, Victoria's agricultural sector has never suffered from an acute lack of space and dense settlement comparable with that in many other places (e.g. the Netherlands). All such comparisons are influenced also by historical and statistical accident: it is probably possible to delineate areas in North America or the Baltic of comparable size to Victoria that retain a great deal of coastal marsh, just as it is possible to select large areas in Victoria that show heavy losses. Added to this is the likelihood that studies reporting depletion tend to focus on areas under threat, rather than broader administrative units such as individual States or Territories. It is also important to note that the coastal marshes surveyed in this study do not tell the whole story about the loss of wetland ecosystems across the State. For example, near-coastal freshwater wetlands (often dominated by Swamp Paperbark *Melaleuca ericifolia*) have suffered enormous losses to agriculture, and they are not reflected in our analysis (East 1935, Callaghan 1948, Roberts 1985, Yugovic & Mitchell 2006).

In those places where losses have been significant, a large number of reasons can be outlined as being responsible (e.g. see reviews by Laegdsgaard 2006; Laegdsgaard *et al.* 2009; Thomsen *et al.* 2009). In some areas of Victoria (e.g. northern and eastern Port Phillip, and the western shore of Western Port), the creation of a large metropolis with associated port, industrial and residential zones is undoubtedly responsible. In many others, however, substantial losses have occurred in areas that have experienced little direct population pressure (e.g. Anderson Inlet, Shallow Inlet, and Corner Inlet). In these areas, drainage and infilling for agriculture have probably been responsible. In other cases, extraction of shell-grit, conversion into salt-making ponds, and burning for barilla production has been responsible for the loss of large areas of coastal wetland, as noted in the Introduction.

Ecological implications of the loss of coastal marshes

There is increasing concern across the world about the loss of coastal wetlands and, in temperate areas, of saltmarshes in particular (e.g. see reviews by Teal & Howes 2000, Kennish 2001, Adam 2002, Cattrijsse & Hampel 2006, Doody 2008, Callaway & Zedler 2009, Turner 2009). Silliman *et al.* (2009, page 391) went as far as to say 'salt marshes are under siege from human disturbance on a global scale. Salt marsh coverage as well as the structure of these ecosystems continues to deteriorate drastically due to human-induced changes. The critical ecosystem services these systems support are likewise endangered'.

Much of the concern arises because coastal wetlands are critical habitats for a wide range and large number of birds (Stevens *et al.* 2006, Faunch & Serafy 2006, Wilson *et al.* 2007, Spencer *et al.* 2009, Visser & Baltz 2009). In Victoria, coastal saltmarsh provides the overwintering foraging sites for the critically endangered Orange-bellied Parrot *Neophema chrysogaster* (Loyn *et al.* 1986). There is now growing evidence that the loss of coastal saltmarsh can be tied directly to population declines in some bird species (e.g. see Ganter *et al.* 1997, Wilson *et al.* 2007, Robledano *et al.* 2010, Jenner *et al.* 2011). Coastal wetlands have also long been known as critical sources of organic matter for adjacent estuarine waterbodies, although the mechanism by which the two interact is unclear (Teal & Howes 2000, Zimmerman *et al.* 2000, Kreeger & Newell 2000). Recent research, however, lends support to the notion that coastal saltmarsh is particularly important in terms of providing habitat for small invertebrates and fish, which then move into the estuary on out-going tides or when they are mature (e.g. see Stevens *et al.* 2006 for fish; Mazumder *et al.* 2006 for crabs), rather than supporting off-shore food webs by exporting vascular plant detritus. Thus it can be expected that the loss of coastal wetlands would have serious implications for estuarine food webs (e.g. see Svensson *et al.* 2007).

Conclusions

Our study documents the loss of coastal marsh across the State of Victoria from ~1835 to 2008. We believe it to be the first such study of this scale in Australia. Its scope reveals large regional differences between sectors of Victoria's coast, but this scale undoubtedly also glosses over many small changes, each with important impacts and interesting stories. It is hoped that the study encourages local studies in areas of particular interest or concern, and in this the recently available LiDAR data for the entire State will undoubtedly prove very useful.

Changes to coastal marshes have not stopped and are unlikely to cease in the near future. The destruction of coastal wetlands for industrial and port development remains an ongoing threat in many parts of Victoria (e.g. Western Port). A more widespread threat, of course, is that posed by climate change. In the absence of sufficiently rapid increases in sediment elevation (Morris *et al*. 2002, Rogers *et al*. 2006), rising sea levels will force coastal marshes to retreat upslope, a response which is impossible in many cases due to topography or human infrastructure (Gilman *et al*. 2008, Boon *et al*. 2010). Clearly, this is not the end of the story for the depletion of coastal marshes in south-eastern Australia.

Acknowledgements

This study, and the Victorian Saltmarsh Study (2011) on which is based, used field data collected by Biosis Research and Ecology Australia in order to obtain a State-wide view of Victoria's coastal marshes. We gratefully acknowledge the time and expertise involved in that fieldwork. We also acknowledge the work of Keshia Atchison (La Trobe University) and Michele Kohout (ARI) in drawing map linework, and Matt White (DSE), Jeff Yugovic (Biosis Research), Andrew McMahon, Geoff Carr, Steve Matthews (Ecology Australia) and Doug Frood (Pathways Bushland & Environment) for useful discussions. Judith Scurfiled (SLV) and Quentin Slade (NLA) are thanked for their help in locating survey plans.

> "Since the acceptance of this manuscript, several additional Parish Plans relating to the Gippsland coast have been brought to our attention. The plans have recently become available digitally via the National Library of Australia and/or the State Library of Victoria, and were not retrieved in our initial search. These plans are resolved at a fine scale, and provide interesting local detail and clarification, but do not alter the findings presented here. For the sake of future researchers who may wish to gain a more complete picture of the historical resource, the relevant Parishes are Alberton West, Balloong, Bengworden South, Nuntin, Sale, Tarra Tarra, Toora, Welshpool, Wonga Wonga South and Yanakie."

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